THREATENED STAG BEETLES IN TASMANIA'S PRODUCTION FORESTS: SINGLE-SPECIES STUDIES CONTRIBUTING TO BIODIVERSITY CONSERVATION

J. M. Meggs BSc(Hons)

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requirements for the Degree of

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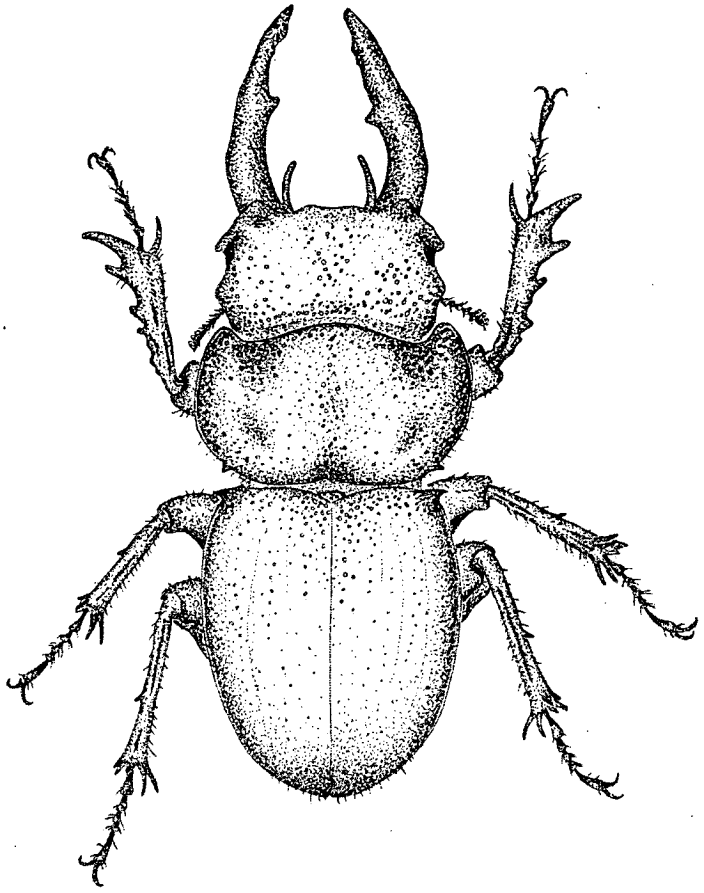

Adult male *Hoplogonus simsoni* Parry (Coleoptera: Lucanidae).
Drawing by Karen Richards.

All parties to the biodiversity debate...must see the larger task - stewardship of all *of the species on* all *of the landscape with every activity we undertake as human beings - a task without spatial and temporal boundaries* (Franklin 1993)

ABSTRACT

Although many formal reservation targets have been attained under Tasmania's Comprehensive, Adequate and Representative (CAR) reserve system, a large proportion of habitats important for threatened species, particularly invertebrates, remain in the off-reserve landscape. The development of conservation management strategies for species whose habitat coincides with areas subject to intensive forestry practices has been severely hampered by a lack of knowledge of the characteristics, spatial distribution and extent of habitats they occupy, and the of the planned disturbance regimes. The distribution, habitat and conservation requirements of three species of stag beetles (Coleoptera: Lucanidae) listed under the schedules of the Tasmanian *Threatened Species Protection Act* 1995, *Hoplogonus simsoni*, *Lissotes latidens*, and *L. menalcas*, were investigated in order to inform the development of management strategies for the conservation of their habitat throughout production forests.

All three species are still restricted to small portions of the State, however this study found that they have wider distributions than previously recorded. Wet eucalypt forest and mixed/rainforest constituted potential habitat for all three species. Through relatively fine-scale predictive habitat modelling it was possible to identify with a reasonable level of confidence the spatial distribution of quality habitat for the litter-dwelling lucanid *H. simsoni*, and the areas where this habitat overlaps most strongly with planned forestry activities. The abundance of *H. simsoni* was related to a particular forest structure, in addition to altitude, aspect and slope. Only broad forest types proved to be a reliable predictor of the occurrence of the obligate log-dwelling lucanid, *L. menalcas*, and the soil-dwelling *L. latidens*. However, as with *H. simsoni*, the abundance of *L. latidens* was associated with a particular forest structure, but also the quantity of coarse woody debris (CWD).

The conversion of potential wet forest habitat to plantation resulted in the local extinction of *H. simsoni*. Given their association with CWD, it is expected that this practice would have a similar effect on the two *Lissotes* sp. Whilst all three species occurred in silvicultural regeneration following clearfelling, there was insufficient evidence to determine the effects of such practices on the long-term viability of

populations of the species. Of concern for all three species is the loss of structural habitat features identified as important to the species through clearfell, burn and sow (CBS) forestry practices. There is an urgent need to determine the ecological sustainability of present forest management practices, such as CBS, in relation to the maintenance of CWD over successive rotations.

All three species were inadequately reserved across their ranges with the bulk of their habitat coinciding with areas managed as production forest. A multi-scaled approach to the conservation of the three species' habitats, incorporating the maintenance of forest structural elements identified as important to each species, is recommended throughout their ranges. In particular, 'off-reserve' conservation strategies for the species need to include limits on the area of potential habitat that may be converted to plantation. The conservation requirements of these three threatened species illustrate the need to expand the focus of habitat conservation from static, set-aside approaches to strategies that incorporate the temporal dynamics of habitat maintenance. The value of single-species studies to the conservation of forest biodiversity in general is discussed.

TABLE OF CONTENTS

ABSTRACT *1*

TABLE OF CONTENTS *3*

GENERAL INTRODUCTION *7*

CHAPTER 1 ..... *11*

THE DISTRIBUTION AND HABITAT CHARACTERISTICS OF A THREATENED LUCANID BEETLE, HOPLOGONUS SIMSONI, IN NORTH-EAST TASMANIA *12*

ABSTRACT 13
INTRODUCTION 14
METHODS 15
Study area 15
Animal survey 16
Habitat variables 19
Statistical analyses 19
RESULTS 21
Distribution and relative abundance of H. simsoni 21
Forest types and the occurrence of H. simsoni 23
Relationship between the occurrence of H. simsoni and measured habitat variables 25
DISCUSSION 28
Conservation considerations 34
ACKNOWLEDGEMENTS 37
REFERENCES 37

CHAPTER 2 ..... *44*

DEVELOPMENT AND EVALUATION OF PREDICTIVE HABITAT MODELS TO ASSIST THE CONSERVATION MANAGEMENT OF A THREATENED LUCANID BEETLE, HOPLOGONUS SIMSONI, IN NORTH-EAST TASMANIA *45*

ABSTRACT 46
INTRODUCTION 47
METHODS 49
Study area, animal survey and habitat variables for model development 49
Model development 50
Predictive map development 50
Field methods for the collection of model evaluation data 52
Model evaluation analyses 52
RESULTS 53
Predicted distribution of H. simsoni and the relationship between its occurrence and measured habitat variables 53
Evaluation of the habitat models 57
DISCUSSION 60
Conservation management considerations 64
Conclusions 68
ACKNOWLEDGEMENTS 69
REFERENCES 69

CHAPTER 3 ..... *74*

DISTRIBUTION, HABITAT CHARACTERISTICS AND CONSERVATION REQUIREMENTS OF A FOREST-DEPENDENT THREATENED INVERTEBRATE LISSOTES LATIDENS (COLEOPTERA: LUCANIDAE) *75*

ABSTRACT 76
INTRODUCTION 77
METHODS 79
Study area 79
Site selection 80
Animal Survey 81
Habitat variables 82
Statistical analyses 83
RESULTS 84
Distribution and relative abundance of Lissotes latidens 84

Relationships between the occurrence of L. latidens and habitat variables 86
Microhabitat characteristics 90
Occurrence of Lissotes latidens and habitat disturbance 91
DISCUSSION 91
Distribution and relative abundance of Lissotes latidens 91
Habitat characteristics 92
Conservation considerations 95
Conclusions 99
ACKNOWLEDGEMENTS 100
REFERENCES 100
CHAPTER 4 ..... *108*
DISTRIBUTION AND CONSERVATION STATUS OF THE MT MANGANA STAG BEETLE LISSOTES MENALCAS (COLEOPTERA: LUCANIDAE) *109*
ABSTRACT 110
INTRODUCTION 110
METHODS 112
RESULTS 114
Distribution 114
Habitat requirements 115
Distribution by land tenure and land use 117
DISCUSSION 119
Distribution 119
Habitat requirements 120
Conservation status 121
ACKNOWLEDGEMENTS 122
REFERENCES 123
GENERAL DISCUSSION *126*
Development of conservation strategies for the H. simsoni, L. latidens and L. menalcas 129
Potential of threatened invertebrates as umbrella species in off-reserve areas 131
Alternatives to the single-species approach to biodiversity conservation 135
ACKNOWLEDGEMENTS *137*
REFERENCES (General Introduction & Discussion) *138*

GENERAL INTRODUCTION

GENERAL INTRODUCTION

Two major impediments have hindered the goal of conserving forest biodiversity through the ecologically sustainable management of our forests. The first of these has been the simplistic belief on both sides of the forest debate that biodiversity is a 'set-aside' issue that can be accommodated in a few, or even many, large ecological reserves (Franklin 1993; Brown 1996; Lindenmayer and Franklin 2002). The second is our limited knowledge of the distribution and ecology of the bulk of forest biodiversity, ie. the invertebrates, leading to a reliance on higher plant communities and large vertebrates as umbrella taxa (Yen and Butcher 1997).

In Tasmania, much of the focus on the conservation of forest-dependent biodiversity has been directed at the expansion of the existing reserve system to achieve a Comprehensive, Adequate and Representative (CAR) reserve system. The most recent example of this, the *Regional Forest Agreement* (Commonwealth of Australia and State of Tasmania 1997), has resulted in a more representative reserve system in terms of forest communities, and the protection of 40% of the State's forest (RPDC 2002). However, it has also resulted in an upsurge of intensive forest management practices in the off-reserve landscape to off-set the loss of timber resource now captured within the reserve system (Forestry Tasmania 1998). As short-range endemism appears to be common in the Tasmanian invertebrate fauna (Mesibov *et al.* 2002), a large proportion of habitats important for many species remain in the off-reserve landscape (Munks *et al.* submitted). Hence, even with an extensive reserve system, such as in Tasmania, significant risks of species loss exist under a strategy that treats biodiversity as a set-aside issue. Off-reserve mechanisms for biodiversity conservation, such as the system of Wildlife Habitat Strips (WHSs) and streamside reserves established throughout production forests in Tasmania (Taylor 1991), are an invaluable supplement to large ecological reserves but do not alleviate the limitations of the set-aside approach, whilst habitat loss for an unknown portion of biodiversity continues in the intervening harvest areas.

If habitat maintenance is the most appropriate method for conserving invertebrates then controlling land-use practices that disturb, alter or damage habitat is probably the most effective way of conserving biodiversity, particularly in off-reserve areas. This

threatening-processes approach has been a significant driver for the compilation of threatened species lists throughout Australia and overseas (Taylor and Bryant 1997). Through its emphasis on threatening processes rather than just rarity, threatened species legislation has potential as an important mechanism for the conservation of habitats important for invertebrates in off-reserve areas. There are currently 99 forest-dependent threatened invertebrates listed under the schedules of the Tasmanian *Threatened Species Protection Act* (TSPA) 1995 with at least a portion of their habitat occurring in areas of production forest (Forest Practices Board 2002). In broad terms, these species represent a range of habitats including headwater and larger streams, seeps, caves, decaying logs, leaf litter and grasslands (Forest Practices Board 2001a).

Since its inception, the TSPA has been implemented with an emphasis on habitat conservation throughout the range of a species rather than simply the protection of known localities of threatened species (Munks and Taylor 2000). For forest-dependent threatened species with habitat that is found in production forests, implementation of the TSPA occurs via the *Forest Practices Act* administered by the Forest Practices Board (Munks and Taylor 2000). Under this Act, forest managers responsible for forestry operations on both public and private land are required to take account of threatened species in the development of a forestry operation plan for a particular area (eg. a logging coupe). Where potential habitat for a listed species is present, management prescriptions for the maintenance of that habitat must be implemented within the operation area. This may be sufficient for threatened vertebrate species, such as the wedge-tailed eagle, whose distributions, habitat requirements and responses to disturbance are relatively well recorded (Mooney and Holdsworth 1991; Mooney and Taylor 1996). However, with the increased scale and intensity of modern forestry operations, it has become apparent that for many listed invertebrate species a coupe-by-coupe approach to fauna conservation has become unworkable, both in terms of species' conservation and timber resource security. In the north-west of Tasmania, for example, plans for a major expansion of plantation on public native forest coincided with a 'hotspot' of abundance for the threatened land snail, *Tasmaphena lamproides* (Bonham and Taylor 1997). This led to a need to develop a comprehensive conservation management strategy for this forest-dependent species across its range (Regan *et al.* 1999). This strategy incorporates conservation measures at both the landscape and coupe scale, ensuring that the conservation requirements of *T. lamproides* are met and

that forestry operations can continue within the species' range, but in a manner that facilitates the maintenance of this species' habitat at multiple-spatial scales and through time (Forest Practices Board 2001b).

Critical to the development of a conservation strategy for any threatened species is an understanding of the distribution and habitat characteristics of the species, as well as specific threats to the species from land-use practices. For many of the invertebrates listed under the TSPA such detailed information is not available (Taylor and Bryant 1997; Munks and Taylor 2000). Of the 99 forest-dependent listed invertebrates for which forestry has been identified as a threat, detailed distribution and habitat utilisation information is available for only 8 species, including *Tasmaphena lamproides* (Bonham and Taylor 1997), *Tasmanipatus anophthalmus* and *T. barretti* (Mesibov and Ruhberg 1991), *Ooperipatellus cryptus* (Mesibov 1993), *Oreixenica ptunarra* (Neyland 1992), and the burrowing crayfish *Engaeus yabbimunna*, *E. orramakunna*, and *E. spinicaudatus* (Doran and Richards 1996). What we do know is that intensive forestry operations, including practices such as clearfell, burn and sow forestry as currently practiced in the lowland wet eucalypt forests of Tasmania (Forestry Commission 1994), lead to a simplification of structurally rich native forests and the loss or fragmentation of late-successional and old-growth habitats (eg. Noss 1999; Taylor and Doran 2001; Lindenmayer and Franklin 2002). Such practices have been associated with the loss of important invertebrate habitats such as coarse woody debris (eg. Harmon *et al.* 1986; Michaels and Bornemissza 1999; Grove *et al.* 2002; Lindenmayer *et al.* 2002), and the decline or loss of late-successional or old-growth specialist species (eg. Niemela *et al.* 1993; Michaels and McQuillan 1995; Spence *et al.* 1996; Niemela 1997; Michaels 1999). Michaels and Bornemissza (1999) have identified stag beetles (Coleoptera: Lucanidae) as a group that may be particularly vulnerable to current forestry practices such as CBS and conversion of native forest to plantation. Tasmania has a rich and highly endemic lucanid fauna, with 39 species currently described (L. Hill unpubl. data). At least half of these are considered to be CWD-dependent for at least a part of their life cycle (G. Bornemissza pers. comm.). Three species of stag beetles believed to be associated with CWD, *Hoplogonus simsoni* Parry, *Lissotes latidens* Westwood, and *L. menalcas* Westwood, were listed under the schedules of the Tasmanian *Threatened Species Protection Act* 1995 due to their restricted distributions, low population densities and their apparent vulnerability to

modern forestry practices (Jackson and Taylor 1995). All three species were known from only a handful of localities in the highly productive wet forests of the north-east and south of Tasmania, areas targeted for intensive forest management (Forestry Tasmania 1998) and already heavily fragmented by agriculture and existing timber plantations.

The overall aim of this thesis was to identify habitats important to *Hoplogonus simsoni*, *Lissotes latidens* and *L. menalcas* and to identify areas where the conservation requirements of the species may conflict most strongly with proposed forestry operations. The studies presented illustrate the adaptation of techniques, such as predictive habitat modelling, largely developed for vertebrates, to species that are likely to interact with their environment at a much smaller scale. Conservation management strategies developed for the three species as a result of this research are presented and evaluated in terms of the difficulties in determining the precise habitat requirements and responses to disturbance of cryptic species. The value of the 'single-species' approach to the conservation of Tasmania's invertebrate diversity in general is discussed.

This thesis contains four data chapters, presented in the form of manuscripts for publication, but with tables and figures interspersed at relevant points throughout the text. Chapters 1-3 have been submitted to the various journals noted on the title page of each chapter and are currently undergoing peer review. Chapter 4 was published in 1999, but has been reformatted to match the layout of the preceding chapters. A reprint of this paper is included at the end of this thesis.

CHAPTER 1

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THE DISTRIBUTION AND HABITAT CHARACTERISTICS OF A THREATENED LUCANID BEETLE, *HOPLOGONUS SIMSONI*, IN NORTH-EAST TASMANIA

*Jeff M. Meggs[1,2], Sarah A. Munks[1] and Ross Corkrey[3]

[1]Forest Practices Board, 30 Patrick Street, Hobart, 7000, Tasmania.

[2]Present address, Forestry Tasmania, 79 Melville Street, Hobart, 7000, Tasmania.

[3] PO Box 491, Wallsend, 2287, New South Wales.

*To whom correspondence should be addressed: jeffrey.meggs@forestrytas.com.au

ABSTRACT

The distribution and characteristics of habitat utilised by a threatened species of stag beetle in north-east Tasmania, *Hoplogonus simsoni* (Coleoptera: Lucanidae), were examined as a first step in the development of conservation management objectives for the species. The beetle was found to have a restricted distribution of 250 km^2 and its regional distribution appeared to be related to the occurrence of granitic geology and a moderately high rainfall at low elevations. The species was patchily distributed across its range. High-density populations of the species were restricted to the eastern part of its distribution, but over most of its range it occurred at very low densities. Relative abundances of *H. simsoni* were greatest in wet eucalypt forest, with significantly fewer beetles found in mixed forest and rainforest. Dry eucalypt forest was found to be unsuitable habitat although the beetle was found to occur in the ecotone between wet and dry eucalypt forest. Potential wet forest habitat for the species is estimated to encompass 18,200 ha or 72% of its range. The species was not found in areas of wet eucalypt forest that had been converted to pine plantation. However, *H. simsoni* was found to occur in wet eucalypt forest regenerating after clearfelling and some of the highest density populations of the species occurred in 70 year old wet eucalypt forest regenerating following a wildfire. The relationship between various habitat variables and the occurrence of the beetle was investigated using Generalised Additive Modelling and robust regression. The presence of wet eucalypt forest below 300 m altitude; slope less than 5°; a deep leaf litter layer; and a forest structure with a well-developed canopy best explained the occurrence of the beetle. These habitat characteristics probably relate to a requirement for a cool, moist, stable microclimate and an absence of wildfire for some time. The potential habitat of *H. simsoni* as identified in this study is poorly reserved across its range and a high percentage has been identified by the forest industry as having potential for conversion to pine plantation. This highlights the importance of having mechanisms for 'off-reserve' conservation of threatened species, like *H. simsoni*, which are often poorly represented in or completely absent from formal reserves.

Keywords: Lucanid beetles; threatened species; forestry; plantations; habitat modelling.

INTRODUCTION

Although many formal reservation targets have been attained under Tasmania's Comprehensive, Adequate and Representative (CAR) reserve system, a large proportion of habitats important for threatened species, particularly invertebrates, remain in the off-reserve landscape (Munks *et al.* submitted). Management objectives and prescriptions to cater for threatened fauna in production forest areas in Tasmania have been developed over the past six years for use under the Tasmanian Forest Practices System (Munks and Taylor 2000; Forest Practices Board 2001). However, for management prescriptions to be effective they must be based on a sound knowledge of the distribution and habitat characteristics of the species, as well as the specific threats to the species from off-reserve land-use practices.

Simsons stag beetle, *Hoplogonus simsoni* (Coleoptera: Lucanidae), is the largest endemic stag beetle in Tasmania and is listed as Vulnerable under the Tasmanian *Threatened Species Protection Act 1995* due to its restricted distribution, low population density and the potential adverse impacts of forestry practices within its range (Jackson and Taylor 1995). With a body length varying between 10 and 24 mm and the greatly enlarged, elongate mandibles of the males, this flightless black beetle is considered the finest of the Tasmanian Lucanidae. It is a relatively long-lived beetle, with the larval stage lasting up to two years and adult lifespan in the range of one to two years (G. Bornemissza pers. comm.). The larvae were believed to be log-dwelling (Greenslade 1985; Jackson and Taylor 1995), however more recent information suggests that they are edaphic forming a part of the diverse soil-dwelling invertebrate fauna (G. Bornemissza pers. comm.; Meggs 1996). The adults are free-living, wandering amongst leaf litter, generally at night, and sheltering under rocks, logs and leaf litter during the day. Meggs (1996) and anecdotal reports from collectors suggest that that levels of activity of adult males and females appear to be highest in December/January for males, and in January/February for females.

Prior to 1996 there had been no detailed studies of *H. simsoni* and the species was known from only five localities in north-east Tasmania (Jackson and Taylor 1995). However, a preliminary investigation of the distribution, habitat and conservation status of *H. simsoni* increased the number of known localities to 18 within a restricted area of

about 300 km^2 in the north-east of Tasmania (Meggs 1996). A moderately high rainfall at low elevations typifies the Blue Tier region in which this species is found, and much of the region is underlain by a granitic geology known as the Blue Tier Batholith (Meggs 1996).

Little of the current known range of *H. simsoni* is contained within formal reserves and much of the forest within its range is of high value to Tasmania's timber industry. Preliminary data suggests that historical practices such as selective logging have had minimal impact on the viability of local populations of the species (Meggs 1996), but little is known of the effects of current forestry practices. Meggs (1996) suggested that the species was patchily distributed, and stressed the importance of identifying key habitats for the species within its range and the need to assess the impacts of habitat disturbance on local populations.

The present study was initiated to provide information for the formulation of management measures required to ensure the conservation of *H. simsoni* in areas subject to production forestry activities. The aims of the study were to determine the distribution and relative population densities of *H. simsoni* throughout its range, determine the characteristics of habitat utilised by *H. simsoni*, and assess the effects of current forestry practices on the beetle. The conservation status of the beetle and conservation management of its habitat is discussed.

METHODS

Study area

All field work for the investigation of habitat characteristics was conducted between November 1996 and May 1997 in an area encompassing the known range of *H. simsoni* in north-east Tasmania (Fig. 1). Additional locality records obtained subsequent to this work have been included to determine the current known range and conservation status of the species but were not used for the analysis of habitat characteristics (Meggs, Munks, and Richards unpubl. data). The forest in the study area consisted predominantly of wet eucalypt forest communities, with some mixed forest and rainforest occurring along riparian corridors and in larger blocks of forest in the centre

and south-western corner of the area. Mixed forest was distinguished from wet eucalypt forest by the presence of rainforest species such as *Nothofagus cunninghamii* (myrtle) and *Atherosperma moschatum* (sassafras) as dominant understorey species. Dry eucalypt forest occurred in the eastern and north-eastern parts of the study area. Approximately 80% of the study area was State forest. Most of the remainder, occurring in the southern half of the study area along the river valleys, was private land and almost 70% of this had been cleared for agriculture in the past 150 years.

The majority of the study area is underlain by a granitic rock-type known as the Devonian Blue Tier Batholith (Groves *et al.* 1977), an extensive composite granitic intrusion of Upper Devonian age covering 1,800 km^2 in north-east Tasmania. It is made up of multiple intrusive bodies or plutons ranging from early mafic granodiorites to late leucocratic granites. The region is subject to relatively high annual rainfall at low elevations. Much of the rain in this area occurs in heavy falls associated with persistent low-pressure systems over the Tasman Sea, which are intensified by the lifting of moist easterly winds up the scarp faces at St. Helens (Mesibov 1988).

The study area forms the north-east corner of an invertebrate bioregion that has been called Plomley's Island (Mesibov 1994 and 1996). This bioregion has been defined by the presence of a number of species of invertebrates that are endemic to Plomley's Island and range widely within it. The eastern and western boundaries of Plomley's Island are fairly distinct faunal breaks where species assemblages change over a relatively short distance (Mesibov 1994). The eastern boundary has been called the Goulds Country Break (Mesibov 1994) and lies within the study area.

Animal survey

Five broad forest types were recognised for the stratification of the survey:

Mature wet eucalypt forest: forest communities of 70 years+ in age dominated by *Eucalyptus regnans*, *E. obliqua* and/or *E. viminalis* with a wet sclerophyll understorey.

Mixed forest/Rainforest: forest communities of 70 years+ in age dominated by *Nothofagus cunninghamii* and/or *Atherosperma moschatum*. Mixed forest was distinguished from wet eucalypt forest by the presence of rainforest species such as

N. cunninghamii and/or *A. moschatum* as dominant understorey species below a wet eucalypt canopy as described above.

Dry eucalypt forest: multiple-aged forest communities, with the oldest cohort of 100 years+ in age dominated by *Eucalyptus obliqua* and/or *E. amygdalina* with a dry sclerophyll understorey.

Clearfelled wet eucalypt forest: wet eucalypt forest communities in coupes which had been subjected to a silvicultural regime of clearfell, burn and sow (Forestry Commission 1994). The ages of the regenerating forest ranged from one to 36 years after logging. Two of the five coupes sampled were less than a year old and had not yet been subjected to a regeneration burn.

Plantation: areas which had previously supported wet eucalypt forest but which had been converted to either pine (*Pinus radiata*) or eucalypt (*Eucalyptus regnans*) plantation.

At least five different geographic locations within each forest type were selected within or immediately adjacent to the range of the species (total of 42 locations). Locations were selected to cover the widest geographic range that would ensure sampling of all combinations of environments within the potential range of the species. At each location for a particular forest type, six sites were selected (total of 252 sites) covering the range of topography (ie, gully/flat, mid-slope, and ridge-top), different aspects, slopes, and proximity to streams, present within a location. Where these attributes were relatively consistent within a location, sites were selected to sample as wide an area as possible. Hence sites were generally located greater than 100 m from one another. Wherever possible sites were located at least 30 m from roads, paddocks or any disturbed habitat not consistent with the forest type under investigation.

Six 1 m^2 plots were placed haphazardly within a 10 m radius circle, thus ensuring all potential microhabitats were sampled, and the plots were systematically searched by hand for live *Hoplogonus* spp. and body parts of dead ones. Any live beetles were recorded and released at the site of capture. Parts of dead beetles were also recorded and stored in 70% alcohol. Identifiable body parts included male heads, female heads

with thorax attached, and the thorax and abdomen of both sexes, which have distinctive humeral spines (Bartolozzi 1996).

Two measures of beetle abundance for each site were estimated from the data: beetle density (no. of individuals/m^2 - calculated from the minimum number of *H. simsoni* known to have been alive from dead parts and live individuals in each plot); and the frequency of occurrence of beetles at each site (calculated as the proportion of plots in which beetles were found).

The hand searching method used in this study may have led to over-estimates of absolute population sizes since the dead parts of beetles sampled may represent an accumulation from more than one active season. A pilot study comparing the abundance estimates for *H. simsoni* produced by pitfall trapping relative to hand-collecting showed that the latter method resulted in estimates three times greater than those obtained by trapping (Wald Statistic = 99.7; df = 1; $p < 0.01$). However, there was no interaction between the method used and location, indicating that hand-collecting was as reliable a method of estimating relative abundance as pitfall trapping. Hand-collecting was thus chosen as the sampling method because of its efficiency relative to trapping and also because it allowed the use of a non-destructive sampling method for this threatened species. Hand-collecting of other litter invertebrates such as millipedes has been found to be more efficient and to provide more accurate data on species relative abundance than pitfall trapping (Mesibov *et al.* 1995).

It was assumed that the presence and abundance of adults reflects that of the soil-dwelling larvae. It may be argued that the larvae of this species are a better indicator of presence, population size and viability, and habitat quality than the adult population. However, although the two life-stages of *H. simsoni* inhabit different microhabitats, the fact that the adults do not appear to eat (P. McQuillan pers. comm.) and are believed to be poor dispersers, in the order of 100-200 m in a life-time (G. Bornemissza and P. McQuillan pers. comms.), supports the assumption that adult presence, abundance and habitat requirements represents the species as a whole.

Habitat variables

Habitat variables recorded at each site were chosen for their anticipated value as predictors of beetle distribution and abundance, and for the ease with which they could be collected. The following habitat variables were assessed at each site: altitude (m), distance to nearest stream (<30 m; 30-100 m; >100 m), leaf litter depth (<1 cm; 1-3 cm; >3 cm), leaf litter cover (% ground cover), rock cover (very low; low; medium; high), dead wood cover (% ground cover of logs >10 cm mid-diameter), moss cover (% ground cover including on rocks and logs), average aspect (N, S, E, W, none), average slope (degrees), distance to nearest road (m), weeds (present/absent).

The forest community species composition and structure was recorded at each site. Forest structure was measured by categorising the forest at each site into vegetation height classes including: overstorey tree, understorey tree, tall shrub, low shrub and ground cover. For each height class present at a site the average height (m) and average canopy cover (%) was visually estimated.

Soil samples were taken from the A1 horizon at each site, placed in a sealed plastic bag, and then dried at the Mt. Pleasant Laboratories (Department of Primary Industries, Water and Environment, Tasmania) for analysis of: percentage organic carbon (Walkley and Black method); total nitrogen (modified from the Kjeldahl method); conductivity (1:5 soil/water extract); pH (1:5 soil/water suspension); and phosphorus and potassium (Colwell bicarbonate-extractable P and K). The procedures for the soil analyses conducted are described in Rayment and Higginson (1992).

Soil profile descriptions were made at eleven sites representing a range of beetle densities. The format of the profile descriptions followed Grant *et al.* (1995).

Statistical analyses

The relative abundances of beetles in each forest type were tested for equality using Generalised Linear Modelling (GLM) (McCullagh and Nelder 1989). The levels of location and site were not included in this GLM analysis as it was merely used to give an overview of differences between forest types as a prelude to the habitat modelling. Variation at the level of location and site was examined using Generalised Linear Mixed Modelling (GLMM). The two measures of relative abundance were used in both

analyses: beetle density (assuming a Poisson distribution and log-link function); and frequency of occurrence (assuming a binomial distribution and logit-link function). Levels of statistical significance were set at 0.01 unless otherwise stated.

Only sites occurring in wet eucalypt forest were used to examine the relationship between beetle occurrence and the measured habitat variables. Plantation and clearfelled wet eucalypt forest sites were excluded because it was decided that the influence of the physical disturbance associated with the clearfelling and associated silvicultural treatments would confound the results. Dry eucalypt forest was excluded because preliminary survey results confirmed the beetle did not prefer this forest type. Mixed/rainforest was excluded due to the small sample size of sites relative to wet eucalypt forest.

Some habitat variables needed logarithm transformation to decrease excessive skewness. These variables are indicated in the results by the prefix 'L'. Initial investigation indicated considerable collinearity between variables within the data, which would have led to biased and poorly estimated model terms (Myers 1990). This was avoided by selecting the model with the fewest significant variables by examination of C_p plots (Snee and Pfeifer 1983). This led to a restricted data set of the following habitat variables: Slope (LSlope), Tall shrub height (TSH), Distance to nearest road, Phosphorus, Altitude (Alt), Overstorey tree cover, Leaf litter depth, Rock cover, Weeds, Aspect, and Distance to nearest stream.

The relationship between the occurrence of *H. simsoni* and the reduced set of habitat variables recorded in wet eucalypt forest was analysed using Generalised Additive Models (GAMs) (Hastie and Tibshirani 1990), incorporating all sites. These are generalised linear models that incorporate smoothed effects that make no assumption on the distribution of the data, and so can be considered non-parametric analyses. Instead of fitting a straight line, a complex curve is fitted. Separate models were constructed for each of the continuous variables crossed with one factor at a time.

Since GAMs cannot be used for predictive models (Hastie and Tibshirani 1990), robust regression using the ROBSSPM procedure in GENSTAT (Payne *et al.* 1993) was used to construct an overall model. It was considered that this approach would lead to a

conservative outcome. Only variables identified as important by the GAMs were included. This procedure does not allow the use of factors and therefore separate models were constructed for each level of the categorical factors.

Separate analyses of variance (ANOVA) and least significance difference (LSD) tests were conducted on the factors identified by the GAM as significantly influencing the abundance of *H. simsoni*. Significance levels were set at 0.05 for both the ANOVA and robust regression.

A multi-dimensional scaling program in the DECODA software package (Minchin 1991) was used to examine the relationship between the structure and species composition of the vegetation communities recorded at each site and the presence/absence of beetles. The Global Non-metric Multi-Dimensional Scaling (GNMDS) procedure was used. At least 20 starting configurations were used in each analysis in two and three dimensions. Minimum acceptable stress levels were set at 0.15.

RESULTS

Distribution and relative abundance of H. simsoni

Hoplogonus simsoni was present in 31 of the 42 locations sampled (119 of the 252 sites sampled). It was found in all forest types with the greatest proportion of new records occurring in wet eucalypt forest (Table 1). During this study a sighting of a *Hoplogonus* specimen was reported from a locality south of the known distribution of *H. simsoni* (Fig. 1). The collector was familiar with the distinctive characteristics of the genus and therefore it is considered a reliable sighting.

Table 1. The percentage of locations and sites sampled for each forest type in which *H. simsoni* was found.

Forest Type	Total locations sampled	Total sites sampled	% of locations with *H. simsoni*	% of sites with *H. simsoni*
Plantation	5	30	20%	7%
Wet eucalypt forest	19	114	95%	62%
Mixed/Rainforest	7	42	71%	43%
Clearfelled wet eucalypt	6	36	83%	50%
Dry eucalypt forest	5	30	40%	17%

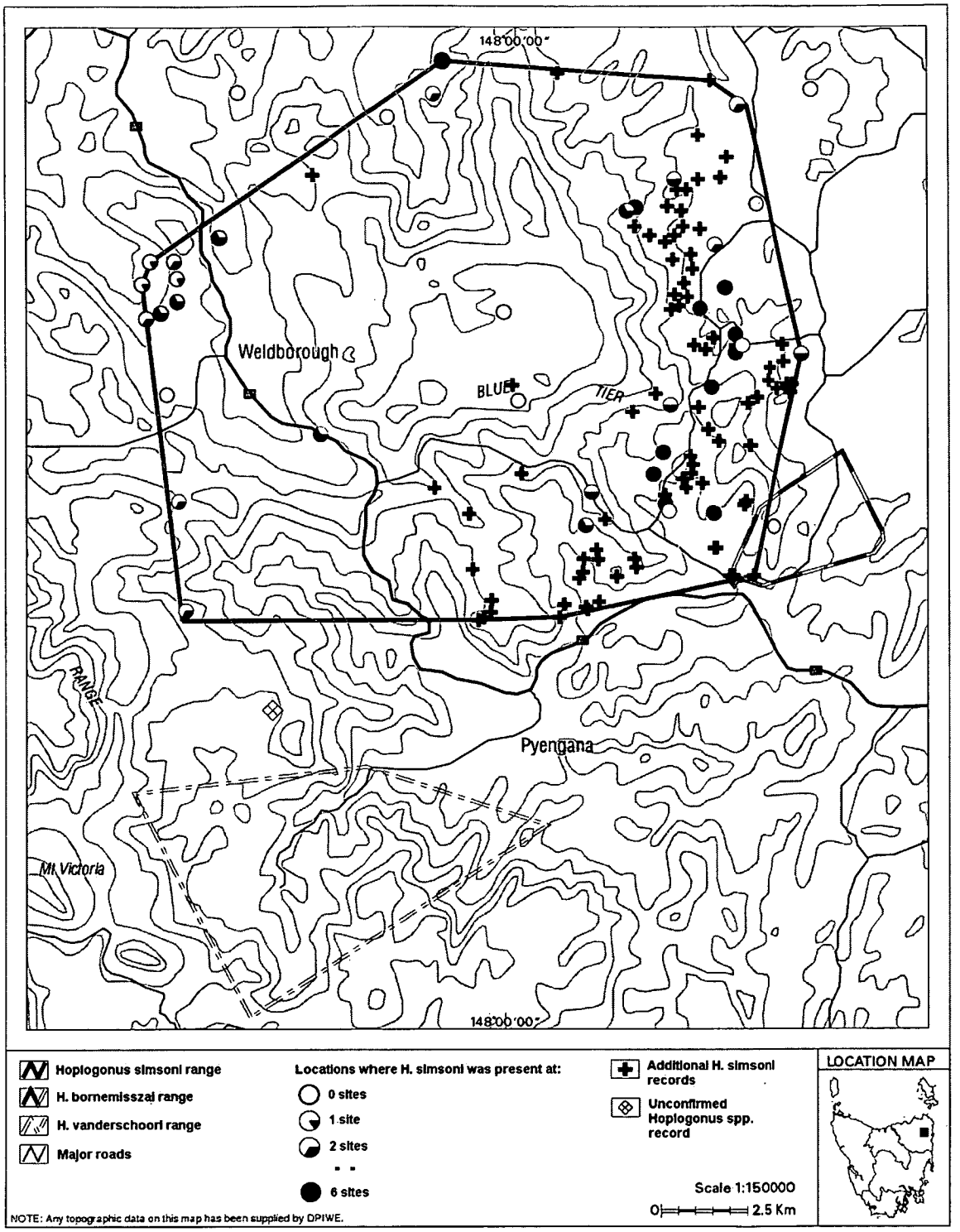

Figure 1. Distribution of *H. simsoni* and the sites surveyed in this study (circles) in north-east Tasmania. The range boundary for *H. simsoni* includes additional records found subsequent to this study (Meggs, Munks, and Richards unpubl. data). Range boundaries for *H. bornemisszai* and *H. vanderschoori* from Munks, Corkrey, Meggs, Richards and Wapstra (unpubl. data).

The previously known range of the species was extended by 10 km^2 in a west/north-westerly direction, by 50 km^2 to the north/north-east and 5 km^2 to the south. The range

was reduced by 15 km^2 in the south-east due to the mistaken identification of a female *H. bornemisszai* specimen by Meggs (1996). Subsequent survey work has further increased the known range of the species by another 10 km^2 to the north/north-east (Meggs, Munks, and Richards unpubl. data).

The extent of occurrence of *H. simsoni*, calculated as the area contained within the shortest boundary that encompasses all known sites of occurrence (IUCN Species Survival Commission 1994) is now over 250 km^2 (25,300 ha) (Fig. 1). This represents an increase of approximately 60 km^2 from that reported by Meggs (1996), recalculated at the 1 km grid square scale as 190 km^2.

H. simsoni was found to occur at very high densities (> 5/m^2) at only one location, in high densities (3-5/m^2) at three locations and in moderate densities (1-3/m^2) at four locations, but over most of its range it occurred at less than one per square metre. Densities varied significantly between locations within forest types (Variance component >> 1; df = 33) and between sites within locations (Var. comp. >> 1; df = 213).

The beetle was patchily distributed between locations within the forest types (Var. comp. >> 1; df = 33). It also appeared to be patchily distributed between sites within locations, but this was not statistically significant, occurring in less than 20% of plots at roughly half of the sites where it was found. These sites largely correspond to those where it was found at low densities. It was more evenly distributed within sites and locations where it occurred at moderate to high densities, such as in the eastern part of its distribution. Overall, within locations, beetle distribution was fairly uniform across sites but actual densities suggested high aggregations for particular sites.

***Forest types and the occurrence of* H. simsoni**

Hoplogonus simsoni was found in all of the forest types sampled (Fig. 2). However, abundance of *H. simsoni* differed significantly between the five forest types for both abundance measures, density (χ_4 = 231, $p < 0.01$) and frequency of occurrence (χ_4 = 51.3, $p < 0.01$). Relative abundances of beetles were highest in wet eucalypt forest and lowest in plantation (Figs. 2a and b).

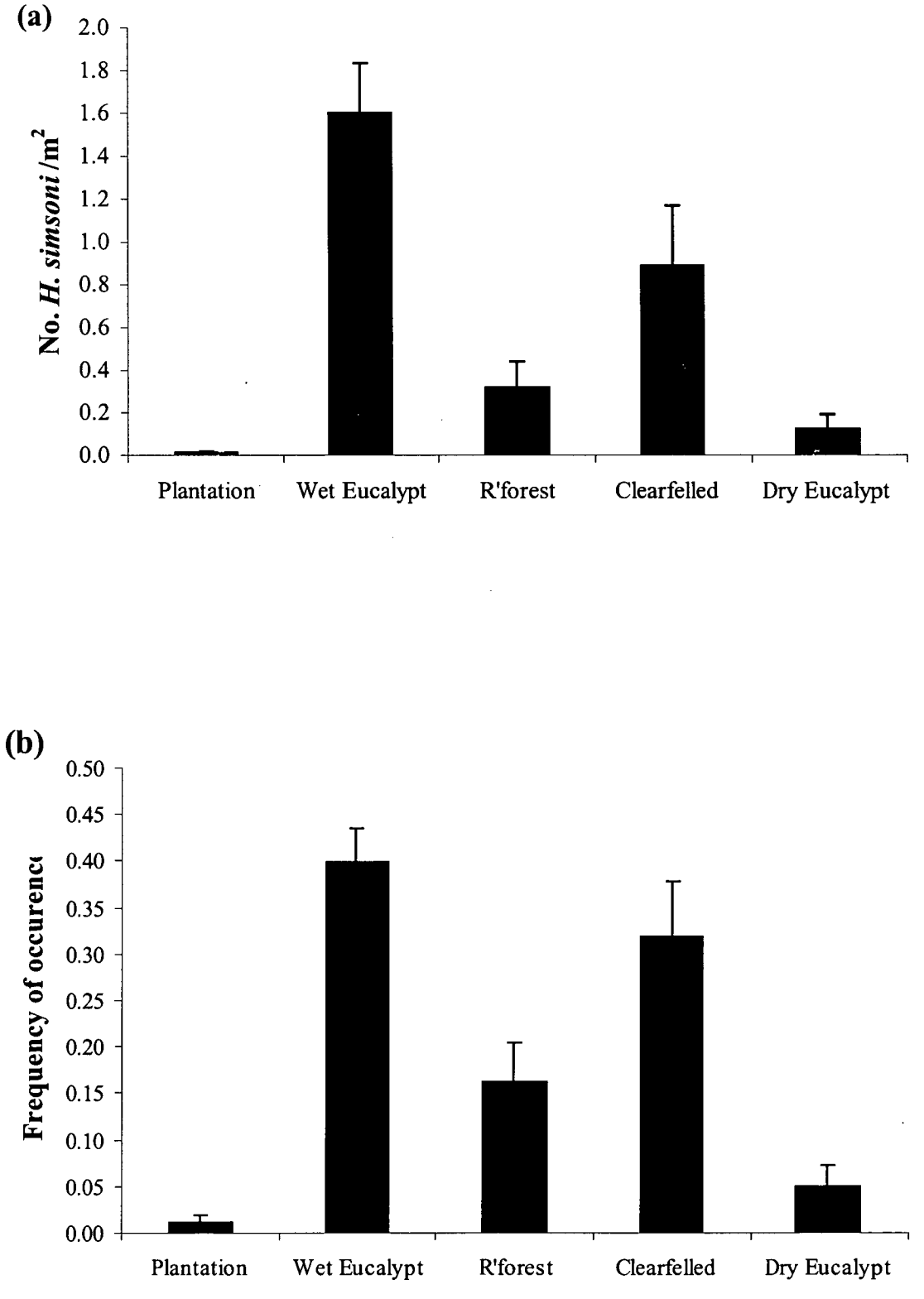

Figure 2. The (a) density and (b) frequency of occurrence of *H. simsoni* in each of the five forest types (mean ± se; n = 30, 114, 42, 36, 30).

Two body parts of *H. simsoni* were found in two plots at one site within a 12 year old eucalypt plantation adjacent (within 50-100 m) to mature native eucalypt forest. However, no body parts of *H. simsoni* were found in any of the pine plantation sites sampled, including a site in a 30 year old plantation immediately adjacent to mature wet eucalypt forest that had the highest density of beetles found in the study. Two males were captured in pitfall traps in this 30 year old pine plantation during the preliminary survey but the absence of any body parts in all of the plots hand-searched in this study suggests that these individuals were vagrants from the adjacent native forest.

Beetles were present in five of the six clearfelled sites sampled including two sites in recently logged coupes, and sites in coupes which were regenerating three years, seven years and 36 years after logging. No beetles were found at the sites in 12 year old *Eucalyptus regnans* forest regenerating after logging. This patch of wet eucalypt forest occurred within an area of predominantly dry eucalypt forest just outside the known range of the species and hence it is possible that the species did not occur in the area prior to logging.

Relationship between the occurrence of* H. simsoni *and measured habitat variables

The Generalised Additive Modelling (GAM) of *H. simsoni* habitat indicated that the following seven habitat variables significantly influenced the abundance of beetles in wet eucalypt forest: Altitude, Overstorey tree canopy cover, Distance to nearest road, Tall shrub height, Slope, Weeds, and Leaf litter depth. The robust regression of *H. simsoni* habitat, which was used to develop overall models using variables identified as important by GAM, confirmed the importance of most of these variables in wet eucalypt forest and their relationships with beetle numbers. The exceptions were overstorey tree canopy cover and distance to nearest road, which did not appear in any of the models. The models that best explain the variation in beetle abundance in wet eucalypt forest are presented in Table 2.

Beetle abundance increased with increasing leaf litter depth. Sites with a shallow litter layer (<1 cm) had significantly lower numbers of *H. simsoni* than sites with deeper litter layers ($F_{2,110} = 3.14$; $p = 0.047$). Although on average sites with the deepest leaf litter (> 3 cm) had the greatest numbers of beetles, these were not significantly higher

than those with leaf litter between one and three centimetres. Numbers of *H. simsoni* were also significantly higher at sites where weeds were present ($F_{1,112}$ = 12.68; $p < 0.001$), in the order of three times greater than at sites without weeds.

Table 2. Robust estimates of regression coefficients for *H. simsoni* habitat in wet eucalypt forest (s.e. = standard error; t = the t-statistic, used as a rough guide to test whether each of the factor levels differ from the first level; % Var. accounted for = the percentage of variance in the data set accounted for by each model; L = logarithm transformation; Alt = Altitude; TSH = Tall Shrub Height; TSH2 = TSH*TSH).

Variable	Estimate	s.e.	t(*)	p-value	%Var. accounted for
Leaf litter and weeds excluded			t(97)		
Constant	38.6700	4.0400	9.57	<0.001	
Alt	-0.0629	0.0105	-5.99	<0.001	
TSH	-0.7830	0.2650	-2.96	0.004	
LSlope	-2.2700	1.0000	-2.27	0.025	
No. beetles = 38.67 - 0.0629×Altitude - 0.783×Tall Shrub Height - 2.27×LSlope					37.3%
Leaf litter < 1 cm & weeds excluded			t(10)		
Constant	1.795	0.291	6.16	<0.001	
LSlope	-0.452	0.136	-3.32	0.008	
No. beetles = 1.795 - 0.452×LSlope					47.7%
Leaf litter 1-3 cm & weeds excluded			t(59)		
Constant	40.1700	4.2500	9.46	<0.001	
TSH	-3.1480	0.7670	-4.10	<0.001	
Alt	-0.0413	0.0108	-3.84	<0.001	
LSlope	-2.4300	1.0000	-2.43	0.018	
TSH2	0.1326	0.0553	2.40	0.020	
No. beetles = 40.17 - 0.0413×Alt - 3.148×TSH - 2.43×LSlope + 0.1326×TSH2					52.9%
Leaf litter > 3 cm & weeds excluded			t(22)		
Constant	27.2400	6.7800	4.02	<0.001	
Alt	-0.0548	0.0230	-2.38	0.026	
No. beetles = 27.24 - 0.0548×Altitude					16.9%
Weeds included & leaf litter excluded			t(9)		
Constant	61.4000	15.1000	4.05	0.003	
Alt	-0.1371	0.0525	-2.61	0.028	
No. beetles = 61.4 - 01371×Altitude					36.8%

The abundance of beetles decreased with increasing altitude and increasing slope. However, at sites with a shallow litter layer and with weeds present the decline of beetle abundance with increasing slope was less pronounced. Abundance of beetles also declined with increasing distance from a road, however this trend was not apparent where litter layers were shallow.

There was a complicated relationship between tall shrub height and beetle abundance. Numbers of beetles declined with increasing tall shrub height but the graph shows a

bowl-shaped curve (Figs. 3a and b). This relationship differed for sites with weeds compared to those without (Fig. 3b).

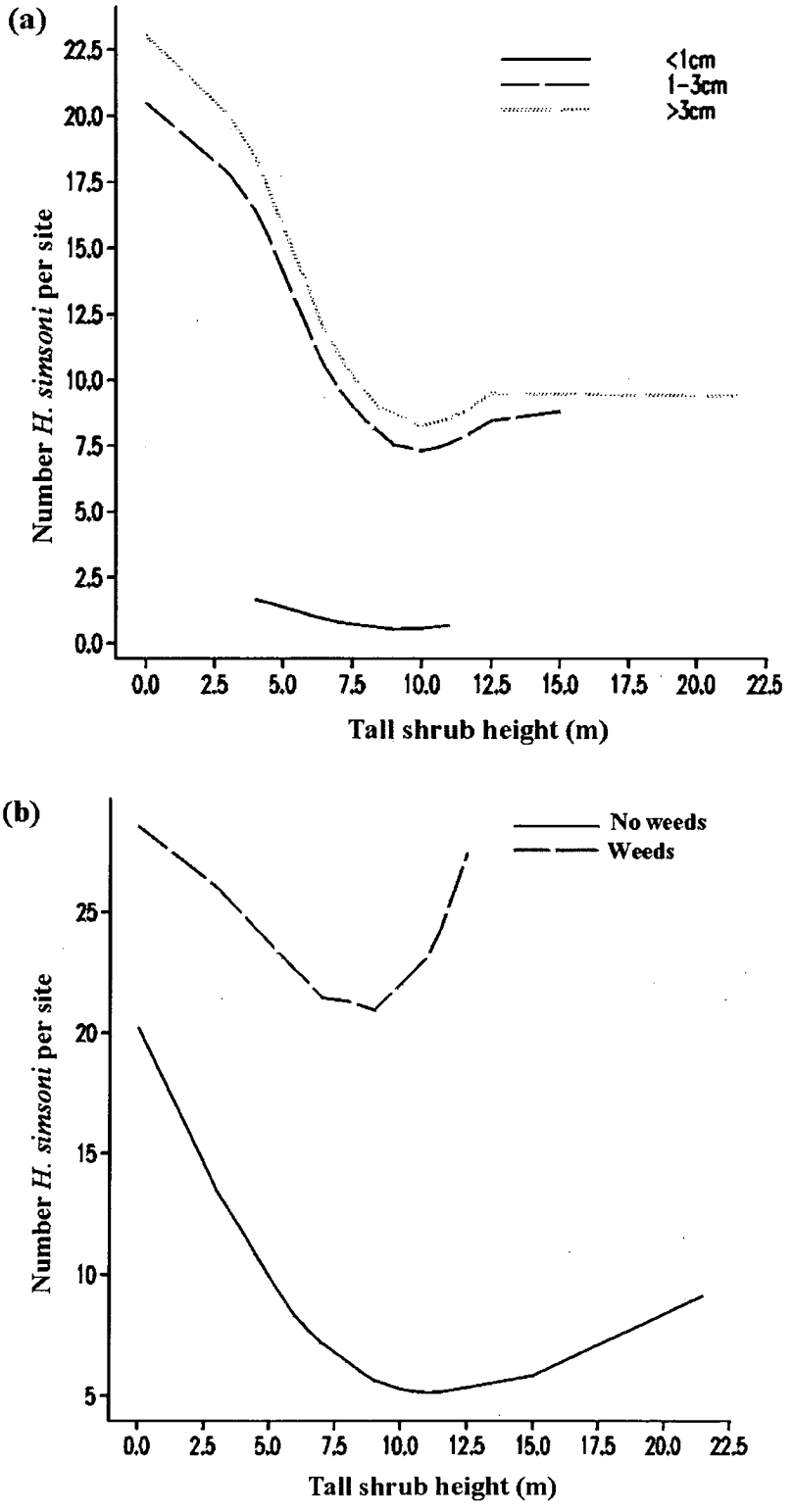

Figure 3. The relationship between numbers of *H. simsoni* per site and tall shrub height for (a) varying leaf litter depth and (b) sites with and without weeds in wet eucalypt forest from the GAM analysis. The fitted lines are smooth curves obtained from the Generalized Additive Models and indicate trends and non-linearities in the data. The length of the lines corresponds to the range of the data collected.

The GNMDS analysis found no clear relationship between plant community species composition and the presence or absence of *H. simsoni*, neither was the presence of *H. simsoni* related to a particular forest structure.

DISCUSSION

The results from this study have shown that *H. simsoni* is more widely distributed in the north-east of Tasmania than was thought when it was first listed as threatened (Meggs 1996). It was recorded from 38 localities in north-east Tasmania, which with the addition of recent records (Meggs, Munks, and Richards unpubl. data) results in a range of roughly 250 km^2, centred on the Blue Tier (Fig. 1), of which 182 km^2 is considered potential habitat. Although small extensions of its range beyond that found by Meggs (1996) have been made, it is unlikely that future searches will extend its range any more than a few kilometres beyond that presented here. Much of its current known range is surrounded by unsuitable habitat (eg. dry eucalypt forest, altitudinal barriers). Where potential habitat does exist, studies on other litter invertebrates have failed to find *H. simsoni* (eg. Bashford 1990; Mesibov and Ruhberg 1991; Bonham 1996; Meggs, 1996; Taylor unpubl. data). The occurrence of two new species of *Hoplogonus* (*H. bornemisszai* and *H. vanderschoori*) on the south-west and east/south-east boundaries of *H. simsoni*'s range (Richards 1999) supports the argument for the south-west and east/south-east boundaries of *H. simsoni* range. Apart from a minor overlap in the distributions of *H. simsoni* and *H. bornemisszai* in a small part of the boundary (Marguerita ridge) the species have not been found to co-occur (Fig. 1), yet all three species appear to inhabit forests of the same type (Richards 1999). In addition, the east/south-east range boundaries of *Hoplogonus* species coincide with part of an invertebrate faunal boundary, respected by at least eight species of invertebrates with minimal overlap, called the Goulds Country Break (Mesibov 1994). This boundary may be explained at an ecological level by the presence of a steep environmental gradient but its maintenance at a species level is unknown (R. Mesibov pers. comm.).

Hoplogonus simsoni occurred at relatively high densities at only a handful of locations. Over most of its range it occurred at very low densities. Most species exist as metapopulations, since habitat heterogeneity and breeding systems divide them into a

series of populations that interact by dispersal between them (Caughley and Gunn 1996). The classic metapopulation structure is characterised by local extinctions and recolonisations. This may take the form of patchy populations in localised and ephemeral habitats, characterised by high rates of individual dispersal between patches, which lessens the chance of local extinctions. At the other end of a continuum of metapopulation patterns is the mainland-island population structure where a large core population supplies immigrants to recolonise islands as local extinctions occur. In contrast to this structure is the source-sink structure where it is not the core population size but habitat quality that determines metapopulation structure. The source population occupies high quality habitat that supplies dispersers to recolonise sink populations in patches of poor quality habitat (eg. Ehrlich and Murphy 1987; Mladenoff *et al.* 1995). Although there were areas within the range of the beetle which were not sampled in this study and hence make it difficult to be certain of the overall structure of its population, the results suggest that *H. simsoni* does have at least one, or possibly two, mainland or source populations. Only four locations of high beetle density (> 3/m^2) were found in this study and they occurred within close proximity of one another in the south-eastern corner of its range (ie, along the southern end of Murdochs Road). Approximately 10 km to the north of this area, near the intersection of Murdochs Road and New England Road, is a grouping of moderate density (1-3/m^2) locations (Fig. 1). However, over the rest of the beetle's range it occurred in low numbers. The beetle's dispersal abilities are believed to be limited, in the order of 100-200 m for an individual in its life-time (G. Bornemissza and P. McQuillan pers. comms.). Therefore, it is probable that the northern and western populations are the most vulnerable to habitat disturbance or environmental change, since they lack any high-density populations nearby to supply re-colonisers when local populations go extinct.

All but seven of the locations where the beetle was found to occur in this study were on a granitic rock known as the Devonian Blue Tier Batholith. This batholith is an extensive composite granitic intrusion of Upper Devonian age covering 1,800 km^2 in north-east Tasmania (Groves *et al.* 1977). With a range of only 250 km^2 the beetle only occupies a small portion of this batholith and therefore this geology alone is clearly not indicative of the beetle's presence. It is likely that a complex relationship between this geology, topography and the amount and pattern of occurrence of the rainfall particular to the Blue Tier area, as described by Meggs (1996), is influencing the regional

distribution of *H. simsoni*. In terms of broad forest types, *H. simsoni* showed a preference for wet eucalypt forest and species abundance was clearly influenced by altitude, slope, leaf litter depth and a specific forest structure. Specifically, the beetle occurred in highest numbers in areas within wet eucalypt forest where the altitude is below 300 m, slope is less than 5°, there is a deep litter layer (at least 1-3 cm) and a well developed upper understorey layer (ie, tall shrub layer).

The majority of these characteristics may relate to the beetle's requirement for a relatively cool, moist, stable micro-climate. The preference for a particular forest structure and a deep litter layer indicates that the beetle prefers relatively mature forest that has not been subject to major disturbance, such as severe wildfire, at least for 50 years but possibly longer. Studies of other forest-dwelling ground beetles in evergreen forest have also found that a particular forest structure influences the occurrence of different species (Niemela and Spence 1994; Ings and Hartley 1999). Leaf litter is likely to be the major source of soil organic matter for *H. simsoni* larvae and therefore ultimately its major food source. Ashton (1975) has shown that in *Eucalyptus regnans* forest the humified layer of leaf litter, which may be mixed with top-soil to a depth of 30 cm, can contain 56% organic matter compared with 12% for normal soil at the same depth. In addition to being a potential food source a deep layer of litter may also maintain a cool and moist microclimate for both *H. simsoni* larvae and adults and provide refuge for adult beetles from predators.

The lack of a relationship between the soil chemical or physical characteristics measured in this study and the abundance of *H. simsoni* is surprising given that larval *H. simsoni* are soil-dwellers and are apparently the only life-stage that feeds (P. McQuillan pers.comm.). It also contrasts with the results of other studies of ground-beetles which have found that beetle community composition has been related to soil properties (Dufrene 1990; Sanderson *et al.* 1995; Luff *et al.* 1992: Niemela *et al.* 1992). It may be that soil properties such as texture and drainage are more important to *H. simsoni* than soil chemistry. Soil profile descriptions were only made at eleven locations representing a range of beetle densities. At locations of high beetle density the soils were of the Stronach type (Grant *et al.* 1995), characterised by a deep (20-40 cm), dark organic clay loam A_1 horizon, with variable levels of coarse sands and gravels. There was then a gradual boundary to the B_2 horizon of light medium clay. The

absence of mottles in the topsoil on flat ground indicated that these soils are relatively well drained (M. Laffan pers. comm.). In contrast, the soils at locations of low beetle density were often as deep but had a much lighter texture and lacked the coarse sands and gravels. However, these differences were not pronounced and any apparent preference for soils of a particular structure will require more detailed investigation.

The significance of the presence of weed species to the occurrence of the beetle as identified in the habitat modelling is unclear. Beetle numbers were found to be significantly higher when weeds were present. Weeds were present in less than a fifth of sites sampled and generally occurred at very low densities of 1-2% ground cover. It is possible that the presence of weeds is indicative of habitat disturbance such as road construction and may reflect the coincidence between the preferences of land managers for locating roads and the beetle's preferred habitat. Roads have long been recognised as an important vector for weed dispersal. If such an indirect relationship does exist, it would suggest that the beetle can tolerate this type of disturbance at least at the current level of intensity.

The majority of wet and mixed eucalypt forest coupes (averaging 50 ha in size) within the range of *H. simsoni* are currently harvested by clearfelling often followed by a high intensity regeneration burn and aerial sowing - a clearfell, burn and sow or CBS silvicultural regime (Forestry Commission 1994). Current sustainable yield models indicate that the regenerating forest may then be subject to harvesting again at approximately 90-year intervals (Forestry Tasmania 2002). Studies of the effects of clearfelling on invertebrates have generally found that initially a significant reduction in individuals and changes to the community structure occurs, including the complete loss of some species and invasion by new species (Huhta *et al.* 1967; Vlug and Borden 1973; Lenski 1982; Szysko 1991; Niemela *et al.* 1993; Michaels and McQuillan 1995; Taylor *et al.* 2000). These effects are thought to be a result of the direct impacts of clearfelling on soil and leaf litter microhabitats, including increased light reaching the soil surface, greater temperature extremes, and changes in moisture conditions (Huhta *et al.* 1967). In addition, some studies suggest that high intensity burns used to regenerate the forest after clearfelling significantly increase the impact on soil and leaf litter fauna (eg. Huhta *et al.* 1967; Koch and Majer 1980; O'Dowd and Gill 1985; Christensen and Abbott 1989; Moldenke and Lattin 1990; Beaudry *et al.* 1997;

Strehlow *et al.* 2002). Generally species that disappear from areas of forest subject to CBS usually correspond to those classified as mature forest or old-growth specialists (Niemela *et al.* 1993; Michaels and McQuillan 1995). The characteristics of habitat preferred by *H. simsoni* determined in this study suggest that this species is a mature forest specialist, hence it was surprising that the species was found to occur in recently harvested sites. It is possible, however, that the body parts found in the recently harvested sites were a legacy of pre-logging conditions, since chitin may take a number of years to break down and the sites were in areas where the beetle occurred in high numbers prior to harvest (Meggs 1996). In addition, two of the coupes sampled had not yet been burnt. However, it is also possible that sufficient moist refuges (eg. under decayed logs and rocks), demonstrated as important by Madden *et al.* (1976) and Moldenke and Lattin (1990), remained within these coupes enabling survival of individuals.

The regenerative patterns for groups of invertebrates that are impacted by clearfell, burn and sow operations are highly variable (Friend 1995). However, most studies have found that within a period of 20-50 years the numbers of litter and soil invertebrates build up again and in some studies this is linked with the recovery of leaf litter depth (Bornemissza 1969; Springett 1976; Madden *et al.* 1976; Michaels and McQuillan 1995). Madden *et al.* (1976) have suggested that wet forests may need to be over 50 years old before the litter fauna return after clearfelling and burning. Similarly, in a study of the abundance and species composition of soil fauna in managed forests in America, old-growth forest taxa were not present until the new canopy was 20-40 years old (Moldenke and Lattin 1990). The results from this study suggest that 30 year old *Eucalyptus regnans* can sustain a viable population of *H. simsoni*. Beetle parts were also found in three year old and six year old regeneration, but in very low numbers. However, it should be noted that no live beetles were found in any of these coupes. All of them were adjacent to large areas of unlogged wet eucalypt forest where beetles occurred in high densities, circumstances that could enhance the chances of recolonisation of the harvested areas.

As wet eucalypt forest appears to be the preferred habitat of *H. simsoni*, it is clear that the species has evolutionary experience of fire. Wet eucalypt forests are naturally fire-prone, and wildfires of various scales and intensities, with a return interval of 20-400

years (Hickey *et al.* 1999; McCarthy *et al.* 1999; Alcorn *et al.* 2001) are thought to be the main source of large-scale natural disturbance. However, many elements of CBS silviculture are qualitatively different from the effects of wildfire, including the resultant forest age structure and the landscape-scale spatial pattern of disturbance (McCarthy and Burgman 1995; McCarthy and Lindenmayer 1998; McCarthy *et al.* 1999). For this reason, and due to the small number of coupes of different ages sampled in this study, caution should be exercised in drawing conclusions on the recovery pattern of *H. simsoni* following CBS. Without any sampling over time it cannot be established whether populations in harvested areas are in decline, or increasing, or stable (e.g. Koivula 2002; Strehlow *et al.* 2002). However, considering the limited evidence from this study, including the fact that the areas of forest containing high densities of *H. simsoni* largely consisted of 70 year old regrowth from the 1929 wildfires (G. Richardson pers. comm.), it is possible that regenerating forest may attain the characteristics of optimal habitat for the beetle within current planned harvesting rotations of 90 years.

Whilst by no means biological deserts (Bonham *et al.* 2002), it is generally accepted that pine and eucalypt plantations have a lower species diversity of invertebrates compared to native forest (Ahern and Yen 1977; Taylor 1991; Mesibov 2001). The results of this study suggest that replacement of native wet eucalypt forest with exotic pine plantations can result in the local extinction of populations of *H. simsoni*. This is likely to be a consequence of both the intensive nature of plantation forest management, which includes a high level of soil disturbance, and the divergence of these forests' microhabitats from the natural forest habitat of the species, both of which may be exacerbated over successive rotations (15–30 year intervals). As *H. simsoni* larvae appear to inhabit the upper soil layer, regular disturbance during plantation establishment is likely to have a significant cumulative impact on populations of the beetle. The microclimate of single-aged plantation stands lacking in understorey is likely to be much more variable than that found in optimal *H. simsoni* habitat, exposing the beetle to greater extremes of temperature. Of perhaps greatest significance for the beetle is the development of leaf litter dominated by a dense mat of pine needles. Ahern and Yen (1977) attributed a significantly reduced species richness and number of individuals of litter invertebrates, including beetles, in pine plantations in Victoria to the homogenous composition of the plantation leaf litter and the chemical nature of the

pine needles. Although no significant differences in soil chemical or physical properties between pine plantations and native forest were found in this study, a change in the composition of the leaf litter may in the long term affect the quantity and quality of food available for the larvae.

The results of this study were insufficient to determine the suitability of eucalypt plantations for *H. simsoni*. Only one eucalypt plantation was sampled in which two abdomens of *H. simsoni* were found in an area adjacent to mature wet eucalypt forest. It is likely however, that the establishment of eucalypt plantations managed for either pulpwood or sawlog will have a similar negative impact on the species as observed in pine plantations, because the same intensive management practices are used and the microclimate is dramatically altered. Bashford (1990) found no breeding populations of carabid species in six year old eucalypt plantations in north-east Tasmania that were adjacent to undisturbed native forest where carabids occurred. He suggested that the lack of a carabid breeding population was due to the loss of a dense ground cover and thick litter layer, resulting in an increased chance of desiccation and a significant reduction in prey availability. Some information suggests that eucalypt plantations may be less favourable for native litter invertebrate fauna than pine plantations (Bonham *et al.* 2002). Although eucalypt plantations managed for sawlog production may have a lesser impact than eucalypt pulpwood plantations due to their longer rotation times (at least 30 years compared with 15 years), it is unlikely that there is sufficient time or opportunity for the recovery of many of the native forest habitat characteristics important to a mature forest specialist such as *H. simsoni*. Indeed, Baguette and Gerard (1993) found that 10-15 year old spruce plantations had a very depauperate carabid fauna relative to 70 year old plantations. They found that it was only in the oldest plantations, as native understorey complexity increased, that forest specialists began to appear in the invertebrate fauna sampled.

Conservation considerations

Although this study and Meggs (1996) extended the range of *H. simsoni*, it still meets the criteria of a vulnerable taxon under the Tasmanian *Threatened Species Protection Act* 1995 due to its restricted distribution, the inadequate reservation of its preferred habitat, and the likely negative impact of modern forestry practices, particularly the conversion of areas of native forest to pine plantation. Potential habitat for *H. simsoni*

(or the predicted area of occupancy of *H. simsoni*) may be broadly defined as all relatively undisturbed wet forest types (including mixed/rainforest) within the species' range. This was estimated to be 18,200 ha (182 km^2) or 72% of the beetles extent of occurrence (Table 3). Although 33% of potential habitat of *H. simsoni* occurs in Tasmania's CAR reserve system, the majority of this is marginal habitat in the form of mixed forest and rainforest. Wet eucalypt forest, the preferred habitat of the species, is relatively poorly reserved with only 19% (1,700 ha) of this forest type occurring in CAR reserves, equivalent to 9% of the potential habitat available to the species throughout its range. The majority of potential habitat occurs in State forest, with 43% of its total habitat potentially subject to forestry practices, including 56% (5,000 ha) of wet eucalypt forest. A further 12% of potential habitat occurs as private land (Table 3). Thirty-eight percent of potential habitat for *H. simsoni* has high or moderate potential for conversion to pine plantation (Forestry Tasmania, unpublished data). These estimates demonstrate the potential loss of *H. simsoni* habitat that could occur under the current intensification of forest management in the north-east of Tasmania (Forestry Tasmania 1998) in the absence of 'off-reserve' management strategies for this species.

Table 3. The distribution of potential habitat for *H. simsoni* (wet eucalypt forest and mixed/rainforest) within its range according to land-use category prior to any species-specific land-use changes on State forest. Other public land category includes miscellaneous land uses such as HEC land.

Land use	Wet eucalypt forest (ha)	Mixed/rainforest (ha)	Total potential habitat (ha)
Formal reserves	900	3,700	4,600 (25%)
Informal reserves	800	600	1,400 (8%)
State forest (couped)	5,000	2,900	7,900 (43%)
State forest (uncouped)	1,000	900	1,900 (10%)
Other public land	100	100	200 (1%)
Private property	1,200	1,000	2,200 (12%)
Total	9,000	9,200	18,200 (100%)

Source: GIS Section, Forestry Tasmania.

Criteria have been established for determining levels of reservation for forest communities in Australia (JANIS 1997) but there are no similar established criteria for adequate levels of reservation for threatened species. Under the *Guidelines for the listing of species under the Tasmanian Threatened Species Protection Act 1995* (Scientific Advisory Committee 1998) the conservation requirements of nominated fauna species are assessed on a case-by-case basis. It is generally accepted that the protection of threatened fauna should aim for the maintenance of viable populations throughout the species range (ie. 100% conservation) (Grove *et al.* 2002). This does not

necessarily equate with 100% reservation or protecting every individual of the species or every patch of its habitat. The specific conservation actions required to meet this aim depend on the status, degree of threat and range of the species. A variety of conservation management approaches ranging from reservation to management by prescription or the development of species recovery plans may be required to achieve adequate levels of protection.

'Off-reserve' conservation strategies to ensure the maintenance of viable populations of *H. simsoni* across its range need to include limits on the area of potential habitat that may be converted to plantation. Areas of wet eucalypt forest with the characteristics of optimal habitat as identified in this study should be excluded from plantation development. In accordance with the recommendations under the Tasmanian *Regional Forest Agreement* (Commonwealth of Australia and State of Tasmania 1997) for the conservation of the species, the high-density populations of the species in the eastern part of its range should be reserved. Over the rest of the species' range, the long-term survival of *H. simsoni* in areas subject to CBS forestry practices is likely to be largely dependent on emigration from source or mainland populations. Therefore, the retention of contiguous undisturbed areas of forest throughout the beetle's range is important to increase the chance of maintaining genetic exchange between populations and as sources of individuals to recolonise regenerating forest as it becomes suitable. It is important that the species is conserved in all parts of its range in order to maintain genetic diversity as a buffer against long-term environmental change. This could be further facilitated by ensuring the dispersal of harvesting in both space and time in order to allow sufficient time for the regeneration of suitable habitat for *H. simsoni* before an adjacent area is harvested.

The intensification of forestry activities in 'off-reserve' forested areas recently increased following Tasmania's *Regional Forest Agreement* (Commonwealth of Australia and State of Tasmania 1997), resulting in an upsurge in clearing of the native forests in the north-east of the State for conversion to plantations (Munks and McArthur 2001; Lindenmayer and Franklin 2002). At the time of this intensification of forest disturbance there were 20 forest-dependent fauna species (with hydrobiid snails lumped as one group) listed under the Tasmanian *Threatened Species Protection Act* 1995, including *H. simsoni*, which were known to occur in areas identified as having

potential for production forestry activities (Forest Practices Board 1998). The development of management prescriptions to maximise retention of quality habitat and/or minimise adverse effects of disturbance for these species has been severely hampered by a lack of knowledge of the characteristics and extent of habitats they occupy, and the species-specific impacts of the planned disturbance regimes (Munks and Taylor 2000). The information collected in this study provides a basis for predicting the distribution of habitats important to *H. simsoni* and the areas where the conservation of the species would conflict most strongly with planned forestry activities.

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CHAPTER 2

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DEVELOPMENT AND EVALUATION OF PREDICTIVE HABITAT MODELS TO ASSIST THE CONSERVATION MANAGEMENT OF A THREATENED LUCANID BEETLE, *HOPLOGONUS SIMSONI*, IN NORTH-EAST TASMANIA

*Jeff M. Meggs[1,2], Sarah A. Munks[1], Ross Corkrey[3] & Karen Richards[1]

[1]Forest Practices Board, 30 Patrick Street, Hobart, Tasmania.

[2]Present address, Forestry Tasmania, 79 Melville Street, Hobart, Tasmania.

[3] PO Box 491, Wallsend, 2287, New South Wales.

*To whom correspondence should be addressed: jeffrey.meggs@forestrytas.com.au

ABSTRACT

The use of predictive habitat distribution models by land managers in the conservation management of threatened species is increasing. Few models, however, are subsequently field-checked and evaluated. This study evaluates the statistical strength and usefulness for conservation purposes of three predictive habitat models developed for a threatened stag beetle, *Hoplogonus simsoni*, found in the wet eucalypt forests and mixed/rainforests of north-east Tasmania. The relationship between various environmental variables for which spatial (GIS) information was available and the density, frequency of occurrence and presence/absence of the species was investigated using generalised linear modelling. Models developed were coupled with the GIS data to develop maps of predicted occurrence within the species' range, grouped into categories of habitat quality. The models found that altitude, aspect, slope, distance to nearest stream and overstorey tree height were significantly associated with the occurrence of the species. Evaluation of the statistical strength of the models with independent data of species' occurrence collected at 95 sites found that the density model performed poorly with little correlation between predicted and observed densities of the species. The frequency of occurrence model, however, showed a moderate ability to predict both species' abundance and presence/absence. The presence/absence model had a similar discriminatory ability in predicting presence or absence of *H. simsoni*, but also showed some potential as an indirect predictor of species' abundance. Assuming a correlation between relative abundance and habitat quality, the frequency of occurrence predictive model appeared to be the better and more direct discriminator of high quality habitat relative to the other models. A conservation management plan for the species is proposed based on this model. The plan includes reservation of predicted high quality habitat and reduction of threatening forestry practices within areas of predicted medium quality habitat. The value of species' habitat models and the need to evaluate their utility in the development of strategies for the conservation of invertebrates in general are discussed.

Keywords: predictive modelling; habitat modelling; model evaluation; GIS; threatened species; forestry; invertebrates.

INTRODUCTION

With the advent of more powerful statistical tools and Geographic Information Systems (GIS) there has been a growing use of predictive habitat distribution models in the management of fauna and flora of conservation significance. Most often, this technique has been used on "charismatic" vertebrates, particularly mammals such as wolves (Mladenoff *et al.* 1995; Corsi *et al.* 1999), the grizzly bear (Mace *et al.* 1999), the black bear (Van Manen & Pelton 1997; Clevenger *et al.* 2002), lemurs (Smith *et al.* 1997), and marsupials (Munks 1993; Lindenmayer *et al.* 1995; Pearce & Ferrier 2001), but also birds (Pearce & Ferrier 2001), including the golden eagle (Fielding & Haworth 1995). Similar studies have been conducted on reptiles (Pearce & Ferrier 2001), and vascular plants (Pearce & Ferrier 2001), including alpine grasslands (Zimmermann & Kienast 1999) and rare plants (Elith & Burgman 2002).

In contrast, there have been very few attempts at creating spatially explicit predictive models for invertebrates. Exceptions include a study by Fleishman *et al.* (2001) that developed statistically significant models predicting species occurrence for 36 of 56 resident butterflies in the central Great Basin of western North America. Rushton *et al.* (1994) created a relatively coarse-scale predictive distribution model that related the occurrence of carabid beetle species in Britain to various land classification categories. In addition, Ehrlich & Murphy (1987) and Murphy *et al.* (1990) have incorporated models of habitat quality with spatial and temporal metapopulation dynamics to examine the population viability of threatened butterflies. The poor representation of invertebrates in the literature on predictive distribution modelling may be because of the lack of comprehensive distribution data (including absences) and quantitative habitat data from a broad range of environments for invertebrates. Environmental variables that may be important to terrestrial invertebrates such as soil, leaf litter and understorey characteristics may not exist in GIS, or may not have adequate surrogates within GIS (York 1999; although this can also be the case for vertebrates, see Lindenmayer *et al.* 1999). Many environmental variables such as climate and topography that can influence invertebrate distributions may exist in GIS at a scale too coarse to be of use for organisms that often operate at small spatial scales (Elith 2000). The paucity of invertebrate studies of this type is also a likely

consequence of the general lack of attention invertebrate conservation has received from land managers (Yen *et al.* 1990).

To effectively conserve threatened species, knowledge is required of their physiological and ecological requirements and responses to disturbance. In the absence of such information, the development of models that predict the extent and distribution of habitats that threatened species utilise can be invaluable to land managers. The generally high stakes involved in threatened species management (ie, potential species' extinction and/or economically or socially significant land-use changes) dictate that such models should be rigorously evaluated (Fielding & Bell 1997; Manel *et al.* 1999; Guisan & Zimmermann 2000). Some predictive habitat models have been subsequently field-checked and evaluated in terms of their statistical strength and/or their usefulness for conservation purposes (eg. Rushton *et al.* 1994; Fielding & Haworth 1995; Lindenmayer *et al.* 1995; Mladenoff *et al.* 1995; Zimmermann & Kienast 1999; Pearce & Ferrier 2001; Elith & Burgman 2002). However, as well as a lack of empirical data, urgency for action is often a driver for the development of predictive distribution models to assist land-use decisions. Hence, model evaluation may not be undertaken (eg. Munks 1993; Smith *et al.* 1997; Mace *et al.* 1999) or the evaluation involves comparing the predictions against the original data used to develop the model (eg. Bustamante 1997; Clevenger *et al.* 2002), often leading to an overly optimistic assessment of the model's predictive ability.

For many of the invertebrates listed in the schedules of the Tasmanian *Threatened Species Protection Act* 1995 comprehensive distribution and habitat utilisation data are not available (Taylor & Bryant 1997; Munks & Taylor 2000). An exception is Simsons stag beetle, *Hoplogonus simsoni* (Coleoptera: Lucanidae), which is listed as Vulnerable due to its restricted distribution, generally low population densities and the potential adverse impacts of forestry practices within its range (Chapter 1). The species is patchily distributed throughout its 250 km^2 range, centred on the Blue Tier in north-east Tasmania, and potential habitat for the species (ie, wet eucalypt forest and mixed/rainforest) encompasses 18,200 ha of its range (Chapter 1). In Chapter 1 optimal habitat for the species was identified as wet eucalypt forest below 300 m altitude, with a slope less than 5°, a deep leaf-litter layer, and a forest structure with a

well-developed canopy. We suggested that these characteristics relate to the beetle's requirement for a relatively cool, moist, stable microclimate and the absence of disturbance for at least 50 years, but possibly longer. We also found that potential habitat of *H. simsoni* was poorly reserved across its range and a high percentage had been identified by the forest industry as having potential for conversion to pine plantation, a practice that results in the local extinction of the species (Chapter 1). It was recommended that the conservation requirements of the species would best be served by the reservation of areas containing high-density populations, limitation of the area of potential habitat that may be converted to plantation, and the retention of contiguous links of undisturbed forest throughout its range (Chapter 1).

The study presented in Chapter 1 provides a basis for predicting the spatial distribution of habitats important to *H. simsoni* and the areas where the conservation of the species may conflict most strongly with planned forestry activities. In the present study the aim was to develop habitat models from the abundance and habitat variable data collected in Chapter 1 to create maps using a GIS to predict the distribution and abundance of *H. simsoni*. The GIS-generated predictions were then field-checked and the habitat models evaluated in terms of their statistical strength and their utility for the conservation management of *H. simsoni*.

METHODS

Study area, animal survey and habitat variables for model development

The species' abundance and habitat variable data used for the development of the models were the same as described in Chapter 1. Details of the study area, the stratification methods for the field survey, and the collection methods for each of the habitat variables and the beetle are described in detail in Chapter 1. Hence, only a brief outline of the original sampling methodology is provided here. All field work was conducted between November 1996 and May 1997 in an area encompassing the known range of *H. simsoni* in north-east Tasmania (Fig. 1). One hundred and nineteen sites in relatively undisturbed wet eucalypt forest and 36 sites in mixed/rainforest were surveyed. Each site was searched for live *H. simsoni* and body parts of dead *H. simsoni* using the method described in Chapter 1. At each site the

following habitat variables chosen for their value as predictors of beetle distribution (Chapter 1) and for which spatial information was available in Forestry Tasmania's GIS were recorded: Overstorey tree height (m); Distance to stream (< or > 100 m); Altitude (in 10 m intervals); Aspect (divided into N, S, E, W, and None); and Slope (degrees).

Model development

The relationship between the presence and abundance of *H. simsoni* and the measured habitat variables was examined using Generalised Linear Modelling (GLM) (McCullagh & Nelder 1989). Wet eucalypt forest and mixed/rainforest were modelled separately because it was found in Chapter 1 that the abundance and presence of beetles differed between the two, with significantly higher numbers occurring in wet eucalypt forest. Some habitat variables needed logarithm transformation to decrease excessive skewness. These variables are indicated in the results by the prefix 'L'. Models were constructed using the two measures of beetle abundance derived from the animal survey data as described in Chapter 1: beetle density and the frequency of occurrence of beetles at each site. A Poisson distribution and log-link function was assumed for the measure of beetle density and a binomial distribution and logit-link function was assumed for the measure of frequency of occurrence. The probability of beetle presence/absence was also modelled (assuming a binomial distribution and logit-link function). A stepwise-forward fitting process was used in which variables were added to the model if the subsequent deviance change was sufficiently large. A large deviance change was chosen so that all variables that were finally included in the model were highly significant. This had the effect of simplifying the model. Only two-way interactions were fitted, again to keep the model relatively simple. Levels of statistical significance were set at 0.01.

Predictive map development

The models for each measure of beetle occurrence (density, frequency of occurrence and presence/absence) were coupled with GIS data for each 100 m (1 ha) grid square within an area encompassing the range of *H. simsoni* to produce three maps of the predicted distribution and/or abundance of *H. simsoni*. GIS data for the significant habitat variables were extracted for the two broad forest types that constitute potential habitat for the species, wet eucalypt forest and mixed/rainforest. These forest types

were identified from Photographic Interpretation (PI) codes (Stone 1998) that categorise patches of forest according to the height and density of the tree layer, and broad type of understorey. Wet eucalypt forest was assumed to be represented by: PI codes of E+3 and above; E-3 and below if in conjunction with T(W) (*Acacia dealbata*); and T(W) on its own. Mixed/rainforest was defined by PI types representing: eucalypt forest of any height potential category in combination with M (*Nothofagus cunninghamii*), Mr (*N. cunninghamii* regrowth), or T (generally *N. cunninghamii* and/or *Atherosperma moschatum*, but may include *Acacia melanoxylon*) in any order; M, Mr and T alone; and any combination of these in any order. PI codes representing non-forested land, plantation, wet forest regenerating from clearfelling, and dry eucalypt forest were excluded. Dry eucalypt forest was assumed to have a PI height-potential code of E-3 and below that was not accompanied by codes indicating wet forest elements (ie, T, M, T(W), Mr).

Altitude, aspect and slope data for each 100 m grid cell were extracted from a 100 m Digital Elevation Model (DEM). Altitude was measured in intervals of 10 m, representing 5 m below and 5 m above the actual value. Aspect data were taken as the direction of downward slope and simplified as a code representing the categories N, S, E, W, and None. Slope data were measured as the inclination of slope calculated in degrees. Overstorey tree height was obtained from the height-class of the matching PI-code. Distance from the nearest stream was obtained by buffering a drainage coverage, and coding areas within 100 m of a stream or lake. All data that were extracted from vector information were converted into grid. The data that were initially grid were based on a cell size of 25 metres. Where both vector and grid coverages required preparation, this was performed in their original state and then converted into the final grid requirements of 100 metre cell size.

To simplify interpretation of the final maps derived from the three models, predictions for each grid cell were grouped into categories representing High, Medium, Low or No occurrence of the beetle. The groupings were based on the relatively low frequency of cells predicted as high occurrence, and the high frequency of cells predicted as low occurrence. For the density map these categories were $> 3/m^2$ (High), $1\text{-}3/m^2$ (Medium), $< 1/m^2$ (Low), and Absent (or not predicted). The categories for the frequency of occurrence map were 0.5-1.0 (High), 0.2-0.5

(Medium), < 0.2 (Low) and Absent. The predicted probability of beetle presence was grouped into the following categories: 90-100% probability (High), 70-90% (Medium), and 50-70% (Low). Where the predicted probability of presence was less than 50% the beetle was considered to be absent. This is a generally recognised cut-off for predictions of species' presence/absence (eg. Fielding & Haworth 1995; Manel *et al.* 1999; Fleishman *et al.* 2001; Clevenger *et al.* 2002).

Field methods for the collection of model evaluation data

Preliminary analysis of the three models, comparing the predicted values for each grid cell with the original observed field data, indicated that the predictions of the Frequency of Occurrence model best reflected the original abundance data ($r = 0.41$; $p<0.05$; $n = 153$). Therefore, this model was chosen as the basis for the stratification of the field survey to evaluate the models' predictive ability. Approximately 30 new survey sites within each of the high, medium and low categories of frequency of occurrence (excluding predictions of absence) were sampled, 95 sites in total (Fig. 1). Sites were selected to cover as wide a geographic range as possible within each predicted category. Where this was not logistically possible, sites were located at least 100 m from one another. All sites were located at least 30 m from roads, paddocks or any disturbed habitat. At each site, six 1 m^2 plots were placed haphazardly within a 10 m radius circle, ensuring all potential microhabitats were sampled, and the plots were systematically searched by hand for live *H. simsoni* and body parts of dead ones (as per Chapter 1).

Model evaluation analyses

A variety of analyses were used to compare predicted and observed occurrences of *H. simsoni* at the evaluation sites. Chi-square analysis in GENSTAT (Payne *et al.* 1993) was used to compare predicted and observed density and frequency of occurrence of beetles. Another measure of a GLM's predictive ability is the Wilcoxon-Mann-Whitney two-sample rank test (Harrell 2000). This test measures the concordance between predictions and actual presence/absence data. Hence, it was used to compare the predicted and observed probabilities of beetle presence, and the frequency of occurrence (converted to presence/absence data). The Wilcoxon-Mann-Whitney test is equivalent to the area under the "receiver operating characteristic" or ROC curve (Hanley & McNeil 1982). A test statistic of 0.5 indicates random

predictions, whilst a value of 1.0 indicates perfect prediction. A test statistic greater than 0.75 demonstrates model utility. Predicted and observed values for all three measures of beetle occurrence were also compared using *Pearson* correlation.

RESULTS

Predicted distribution of H. simsoni *and the relationship between its occurrence and measured habitat variables*

Figure 1 illustrates the map derived from the relationship between the frequency of occurrence of the beetle and habitat variables measured. This map and those developed from the other measures of beetle occurrence indicated that *H. simsoni* has a "ring" or "doughnut"-shaped distribution, with the large "hole" in the centre corresponding to high altitude areas of the Blue Tier (Fig. 1). High-density populations are predicted in a 5-10 km-wide band in the eastern part of the species' range, with relatively isolated populations in the west and north of its range.

Altitude was a significant predictor of the density, frequency of occurrence and presence/absence of *H. simsoni*, with slope and aspect also found to be significantly influencing the density and frequency of occurrence of *H. simsoni*. The height of the overstorey tree layer and proximity to the nearest stream were also significant predictors of the density of *H. simsoni*.

Relative density model

The model constructed for the density of *H. simsoni* per site for wet eucalypt forest consisted of the following set of variables that together best explained the data collected:

Beetle density = constant + Altitude + Aspect + LSlope + Distance from a stream (RipD) + Overstorey tree height (OSTH) + Alt×Aspect + LSlope×Aspect + OSTH×RipD.

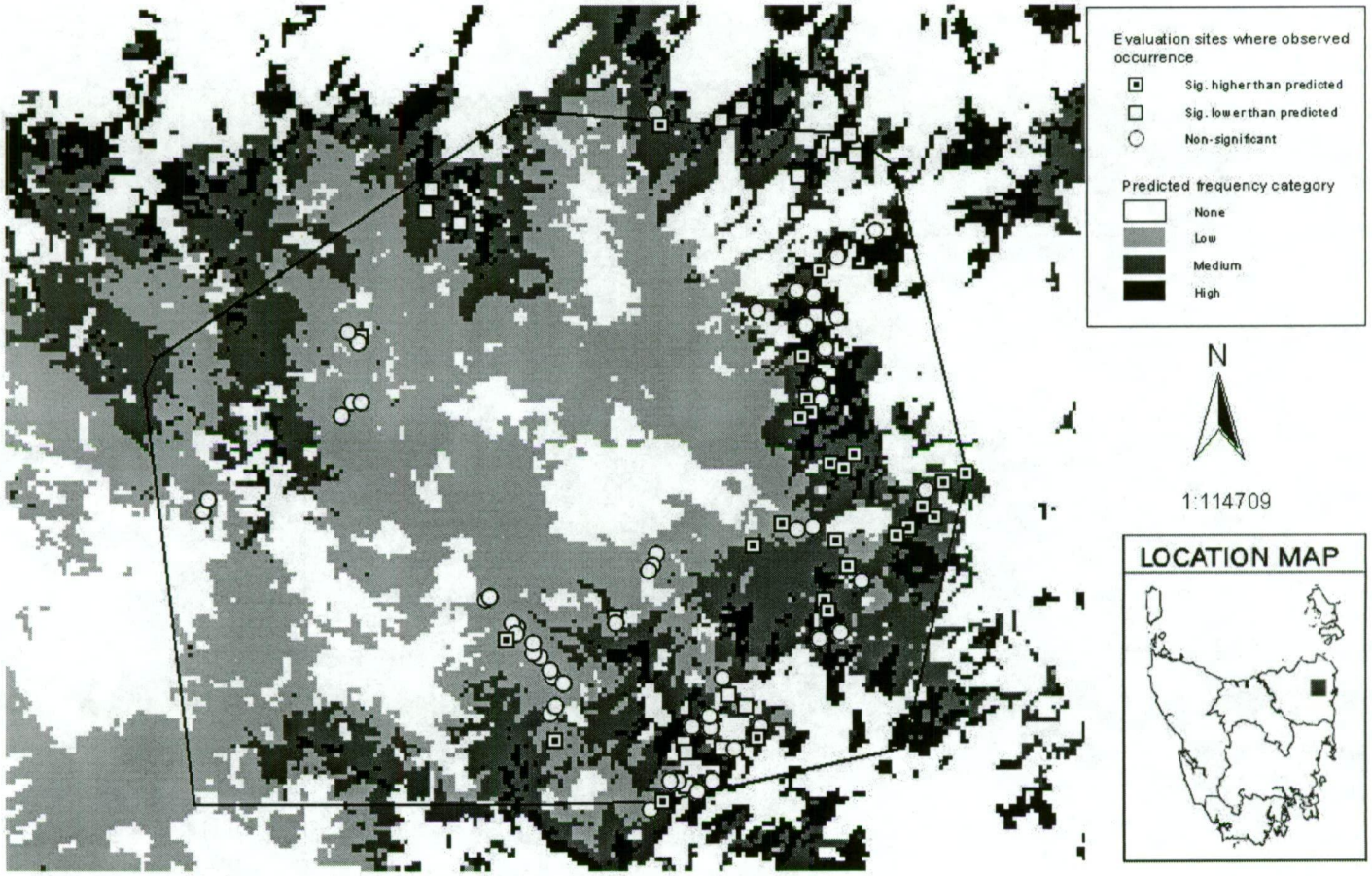

Figure 1. The predicted frequency of occurrence of *H. simsoni* within and adjacent to its 250 km^2 range (solid line) and the location of the evaluation sites, indicating sites where observed values significantly differed from predicted frequencies at $p<0.05$.

The density of *H. simsoni* decreased with increasing altitude at all aspects, but particularly quickly at sites with southerly and northerly aspects (Table 1). It also decreased with increasing slope at northerly and easterly aspects, but there was no significant trend with slope at other aspects. The few sites at which the beetle was found in extraordinarily high numbers were flat and hence had no particular aspect. However, these sites did not appear to exert a strong influence on the overall trends. For sites less than 100 m from streams, beetle density increased significantly with increasing overstorey tree height (Table 1). There was no significant trend at distances greater than 100 m from streams.

Table 1. Estimates of the regression coefficients for the GLM that best explained the density of *H. simsoni* in the wet eucalypt forest and mixed/rainforest surveyed. Significant variables are indicated with an asterisk (s.e. = standard error; t = the t-statistic, used as a rough guide to test whether each of the factor levels differ from the first level. In this case, any t-statistic greater than 2.26 is approximately significant at the 0.01 level.).

Habitat variable	Estimate	s.e.	t (*)
Wet eucalypt forest			
Constant	-0.05400	0.64500	-0.08
Altitude	-0.00165	0.00139	-1.18
Aspect E	0.29100	0.49800	0.58
Aspect S*	1.45400	0.49700	2.93
Aspect N*	3.52900	0.81600	4.32
Aspect None	0.45400	0.56300	0.81
Stream dist. >100 m	5.38700	0.54900	9.81
Altitude X Stream dist >100 m*	-0.01091	0.00123	-8.88
Altitude X E*	0.00482	0.00148	3.25
Altitude X S	-0.00297	0.00163	-1.82
Altitude X N*	-0.00474	0.00196	-2.41
Altitude X Aspect None	0.00117	0.00190	0.62
LSlope	0.12100	0.10800	1.12
LSlope X E*	-0.63500	0.14300	-4.45
LSlope X S	-0.22800	0.13700	-1.66
LSlope X N*	-1.48400	0.23900	-6.22
LSlope X None	-	-	-
Overstorey tree height*	0.06120	0.01130	5.42
Overstorey tree height X Stream dist > 100 m*	-0.05760	0.01220	-4.72
Mixed/rainforest			
Constant	3.432000	0.412000	8.32
Altitude*	-0.004886	0.000655	-7.46
Aspect W*	-0.878000	0.366000	-2.40
Aspect E*	-1.367000	0.376000	-3.63
Aspect N	-0.042000	0.312000	-0.14
Aspect None	-1.627000	0.753000	-2.16

At mixed forest/rainforest sites the significant habitat variables that were influencing the density of *H. simsoni* were:

Beetle density = Constant + Altitude + Aspect

In the mixed forest and rainforest sampled, beetle density again decreased with increasing altitude at all aspects (Table 1). In addition, beetles consistently occurred at higher densities on slopes with southerly and northerly aspects.

Frequency of occurrence model

The habitat variables that had a significant influence on the frequency of occurrence of *H. simsoni* in wet eucalypt forest were:

Frequency of occurrence = Constant + Altitude + Aspect + LSlope

The proportion of plots with *H. simsoni* within a site decreased with increasing altitude at all aspects (Table 2). Beetles occurred with a greater frequency at sites with southerly and easterly aspects although this difference was not significant. There was also a significant trend for the frequency of occurrence of beetles to decrease with increasing slope at all aspects and altitudes.

Table 2. Estimates of the regression coefficients for the GLM that best explained the frequency of occurrence of *H. simsoni* in the wet eucalypt forest surveyed. Significant variables are indicated with an asterisk (s.e. = standard error; t = the t-statistic, used as a rough guide to test whether each of the factor levels differ from the first level. In this case, any t-statistic greater than 2.26 is approximately significant at the 0.01 level.).

Habitat variable	Estimate	s.e.	t (*)
Constant	3.80200	0.53200	7.15
Altitude*	-0.01048	0.00107	-9.80
Aspect E*	0.61500	0.27500	2.23
Aspect S	0.55400	0.25900	2.14
Aspect N*	-1.15500	0.35000	-3.30
Aspect None	-0.33600	0.57800	-0.58
Lslope*	-0.60000	0.15600	-3.85

In the mixed forest and rainforest surveyed only altitude appeared to be influencing the frequency of occurrence of *H. simsoni*. The model took the form:

Frequency of occurrence = 0.512 - 0.004877×Altitude

Again, beetles were found with less frequency as altitude increased (t = -5.46). Sites above 500 m where the beetle was absent appear to have strongly influenced this trend.

Presence/absence model

Altitude was the only habitat variable found to significantly influence the probability of finding *H. simsoni* (ie, the probability of the species' presence) in both wet eucalypt forest and mixed/rainforest. In both forest types the chance of finding the beetle greatly decreased with increasing altitude. The models constructed were:

Prob. beetle presence (wet eucalypt forest) = 3.064 - 0.00827×Altitude

Prob. beetle presence (mixed/rainforest) = 3.620 - 0.00765×Altitude

There was a very high likelihood of finding *H. simsoni* in wet eucalypt forest below 300 m altitude within its range. Sixty-two percent of sites in which the species was found occurred below this altitude compared to only 19% of absent sites. No wet eucalypt forest above 500 m was sampled.

Evaluation of the habitat models

Relative density model

The relative density model proved to be a poor predictor of the abundance of *H. simsoni*. Comparison of the predicted and observed densities of the evaluation sites revealed a Chi-square deviance of almost 30 times greater than would be expected for an acceptable model fit (χ^2_{94} = 2,842). This indicates that the model is over-dispersed, with too much variation at a number of scales for the model to adequately counter random site effects. There was little correlation between predicted and observed densities (r = 0.04). In general, the model appeared to be significantly underestimating relative densities of *H. simsoni* (Fig. 2a).

Frequency of occurrence model

In terms of predicting the species presence or absence the frequency of occurrence model was found to have a moderate discriminatory ability with a Wilcoxon-Mann-Whitney statistic of 0.77. A direct comparison of predicted and observed frequencies of occurrence showed a significant correlation of marginal to moderate strength, with a correlation coefficient of 0.37 ($p<0.05$) (Fig. 2b). Further analysis revealed a Chi-

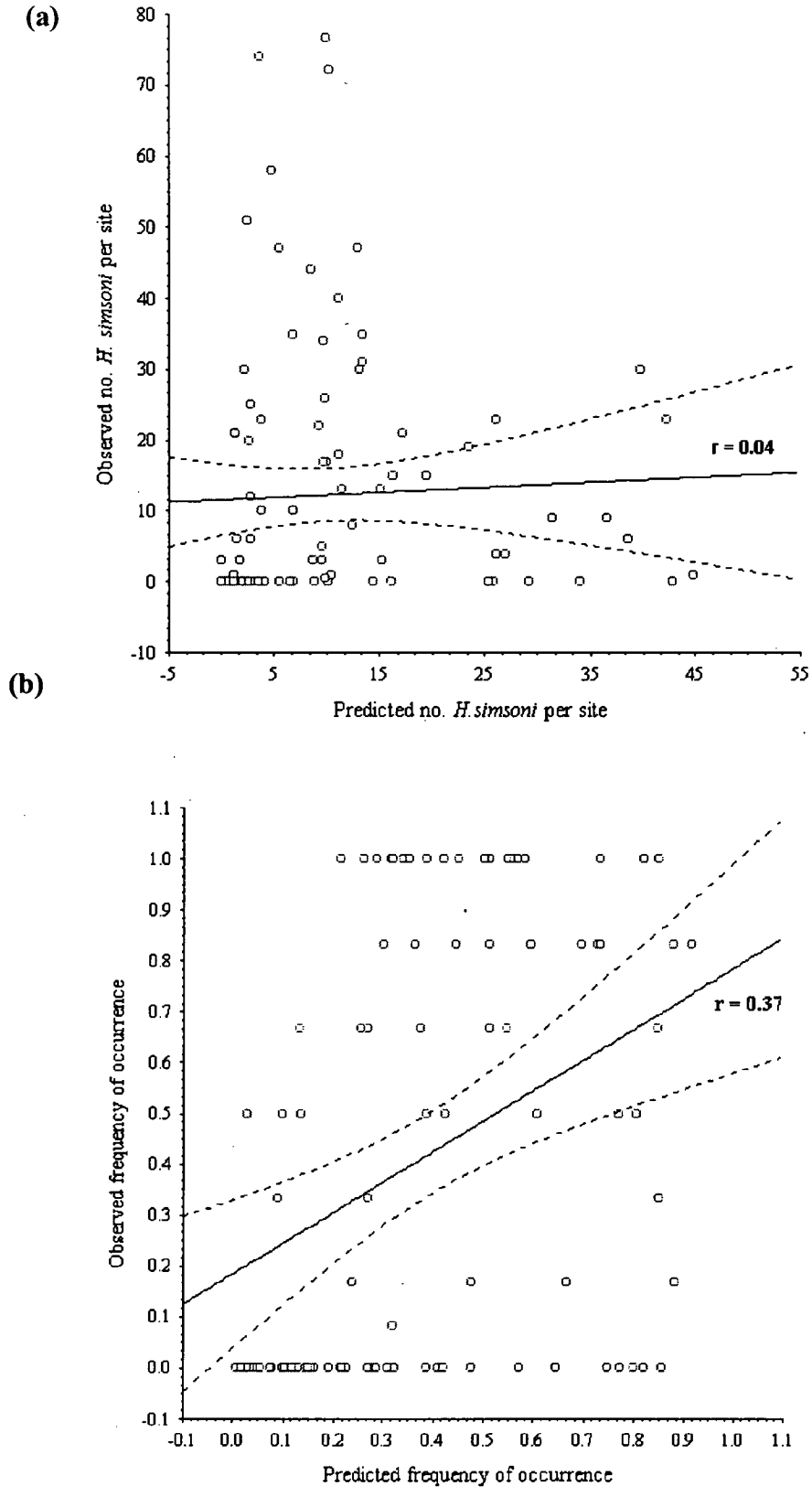

Figure 2. Correlation between the predicted and observed values for **(a)** the density and **(b)** frequency of occurrence of *H. simsoni* at each of the 95 evaluation sites (dotted lines indicate 95% confidence intervals).

square deviance of five times greater than would be expected for an acceptable model fit ($\chi^2_{94} = 512$); indicating that this model was also over-dispersed.

Examination of the spatial distribution of the residuals of the Chi-square analysis (Fig. 1) indicated that the model was significantly over-estimating frequency in the southern and northern margins of the species range, and under-estimating in the largest area of quality habitat in the east of its range.

Presence/absence model

The presence/absence model showed a moderate discriminatory ability in predicting the probability of beetle presence with a Wilcoxon-Mann-Whitney statistic of 0.76, and a significant correlation coefficient of 0.50 ($p<0.05$) (Fig. 2c). A comparison of the predicted probability of presence with observed density ($r = 0.42$; $p<0.05$) and frequency of occurrence ($r = 0.52$; $p<0.05$) indicated that this model had potential as an indirect predictor of species' abundance.

(c)

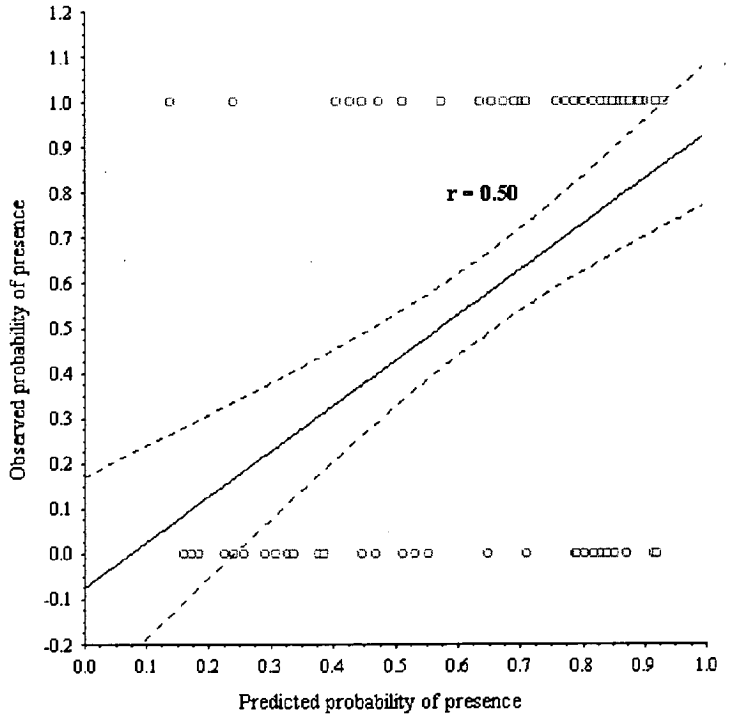

Figure 2. Correlation between the predicted and observed values for **(c)** probability of presence of *H. simsoni* at each of the 95 evaluation sites (dotted lines indicate 95% confidence intervals).

DISCUSSION

It is critical for threatened species management that land managers have access to information on the extent and spatial distribution of habitats important to a species in order to assess the consequences of various land management options. Unfortunately, for most invertebrates, including threatened species, this information is not available. Exceptions include invertebrates with extremely restricted ranges such as the Eltham copper butterfly (Vaughan 1988), or species whose distribution is tied to a particular food plant such as the ptunarra brown butterfly (Neyland 1992) and checkerspot butterflies (Ehrlich & Murphy 1987). However, for more broadly ranging or ecologically opaque invertebrate species, techniques that couple statistical models with GIS may be powerful tools in enabling extrapolation of data of species' distribution and abundance beyond the sites sampled (Nicholls 1989; Neave & Norton 1991). Given the high stakes often involved in threatened species management, the use of an evaluation dataset independent of the model provides the most rigorous test of a model's predictive ability (Chatfield 1995; Fielding & Bell 1997; Guisan & Zimmermann 2000). To our knowledge this is the first study to develop predictive distribution and abundance models for a threatened invertebrate that have then been field-checked and evaluated.

The three predictive models showed variable levels of performance. The density model proved to be a poor predictor of species abundance with little correlation between predicted and observed abundances. It was significantly over-dispersed; a problem consistently found with rigorous tests of GLMs (Pearce & Ferrier 2001). The frequency of occurrence model was also over-dispersed, with too much variation at a number of scales for the model to adequately counter random site effects. The lack of fit of the abundance models may be a consequence of the indirect relationship between the species and the variables modelled, the patchiness of the original survey (Chapter 1), and/or the naturally patchy distribution and abundance of the species (Chapter 1).

Nevertheless, a significant correlation of marginal to moderate strength was found in a comparison of predicted and observed frequencies of occurrence. This model also proved to have a moderate ability to predict the presence/absence of *H. simsoni*. A

simple comparison of the accuracy of the model in predicting categories of species frequency of occurrence revealed that the model was stronger at predicting areas of high abundance. The model was found to accurately predict high abundance for 59% of the evaluation sites predicted as high (Fig. 3). When predictions of high and medium abundance categories were pooled this rose to 68% accuracy. The model performed poorly in predicting low abundance, with the beetle not found at over 80% of sites predicted as low. This may suggest that a minimum threshold of frequency of occurrence should have been applied or that the sampling intensity of the evaluation sites was insufficiently intensive to assess very low predictions of occurrence.

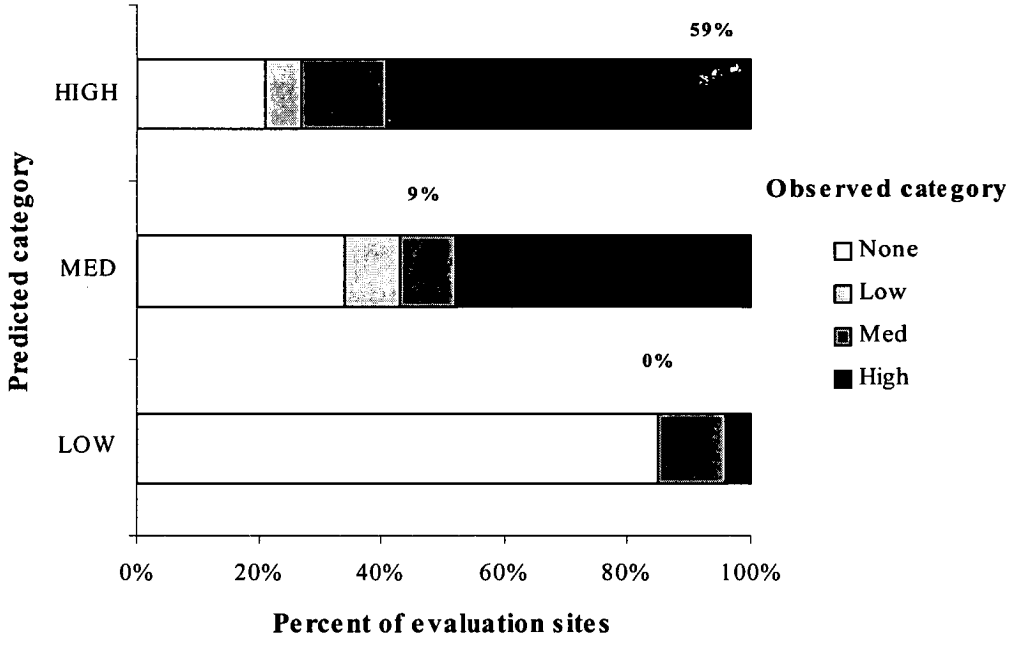

Figure 3. Comparison of predicted and observed categories of frequency of occurrence of *H. simsoni*. The figures on the bars indicate the percentage of sites where the predicted and observed category matched.

The presence/absence model also showed a moderate ability to accurately predict the occurrence of *H. simsoni*. Interestingly, it also exhibited potential as an indirect predictor of species abundance. Pearce and Ferrier (2001) have suggested that more effort should be directed to the collection and modelling of presence/absence data rather than abundance data. They produced reasonably accurate models for 12 of the 44 species of small reptiles, arboreal marsupials, vascular plants and diurnal birds evaluated and found that for all 12 of the species, predictions from direct abundance models performed no better as a relative measure of abundance than predictions from

presence/absence models. The results of this study would appear to support this finding but whether this is the case for other invertebrates requires further investigation.

It is essential that the evaluation of the predictive performance of a species model be done within the context of the original aims of the study (Rykiel 1996; Fielding & Bell 1997). Hence, it has been recommended that a range of criteria be used to evaluate the performance of a model (Manel *et al.* 1999; Guisan & Zimmermann 2000). In this study, we aimed to predict the distribution of habitats important for a threatened species for which we have limited ecological information. Of particular importance was to produce a model to predict the spatial distribution and extent of high-abundance populations of *H. simsoni* and/or optimal habitat (ie, "hotspots" of abundance), as there appeared to be a high correlation between the occurrence of high-density populations and forest with a high potential for plantation establishment (Chapter 1). Whilst the presence/absence model exhibited some potential as an indirect predictor of the abundance of *H. simsoni*, the results of this study suggest that the frequency of occurrence predictive model should be used by land-managers in the development of any conservation management plan for the species. The frequency of occurrence model contains a direct, albeit conservative, measure of species abundance and has shown a moderate ability to predict species abundance. In addition, it was a relatively accurate predictor of areas of high beetle abundance.

This study represents a reasonably successful transfer of conservation biology techniques developed predominantly for vertebrates to an invertebrate of conservation significance. This was unexpected since many predictive distribution and abundance models developed for vertebrates have been plagued by problems of scale (eg. Fielding & Haworth 1995; Lindenmayer *et al.* 1995; Cork & Catling 1996), and invertebrates generally operate on significantly smaller spatial scales. *Hoplogonus simsoni*, for example, is believed to have a dispersal ability of only 100-200 m in an individual's lifetime (G. Bornemissza & P. McQuillan pers. comms.). These problems of scale may be overcome by designing a sampling regime at a scale appropriate to the organism under investigation (Horne & Schneider 1995) and also at a spatial scale at which decisions are to be implemented (Elith 2000). The sampling design used to collect the data from which the predictive models for *H. simsoni* were built

incorporated stratification on the basis of both environmental variability and spatial scale (Chapter 1). Hence, replicate forest types were sampled over as wide a geographic area as logistically possible, and clusters of sites were sampled within each replicate forest type. This allowed measurement of the species' abundance and habitat variability at both a landscape scale (relevant to the scale at which forest management is conducted) and a local scale (relevant to the ecology of the species).

Ecological explanations for the success of variables such as altitude, aspect and slope in predicting the distribution and abundance of *H. simsoni* are uncertain. In Chapter 1 it was suggested that the characteristics of optimal habitat for the beetle related to a requirement of the species for a cool, moist stable microclimate and the absence of disturbance for some time. Altitude and aspect are likely to play a significant role in the local distribution of microclimates. Fleishman *et al.* (2001) found altitude to be the most common predictor in 20 of the 36 significant species distribution models developed for butterflies in the Great Basin of western North America. The importance of gentle slopes to the beetle may relate to the influence of this variable on the development and maintenance of particular soil characteristics. Although based on limited information, in Chapter 1 we suggested that a deep, well-drained soil may be important for the species. These soil characteristics were present at sites with a flat topography and were associated with high beetle densities (J. Meggs, pers. obs.). It has been argued that a successful model used to predict the distribution of a species can serve as a valuable planning tool regardless of whether or not ecological explanations are apparent (Fleishman *et al.* 2001). However, studies have shown that improved modelling success is associated with the use of variables with a known direct or causal effect on a species distribution (Austin & Myers 1995). Establishing causal relationships between the presence and abundance of *H. simsoni* and particular habitat variables or groups of habitat variables remains to be done. The modelling presented in this study and that in Chapter 1 provides a basis for the generation of testable hypotheses.

Most species exist as metapopulations as habitat heterogeneity and breeding systems divide them into a series of populations that interact by dispersal between them (Caughley & Gunn 1996). The predictive map of habitats important to *H. simsoni* generated by the frequency of occurrence model (Fig. 1) appears to support the

contention in Chapter 1 that populations in the east of the species range conform to a metapopulation structure. It appears that *H. simsoni* has at least one but possible two mainland or source populations of high beetle density approximately 10 km apart in the eastern part of its range. Given that the long-term survival of *H. simsoni* in areas subject to clearfell, burn and sow forestry practices is likely to depend on emigration from mainland or source populations (Chapter 1), it is vital that these exceptional populations in the east of its range are protected from large-scale disturbance. The retention of contiguous links of undisturbed forest between these populations and throughout the rest of its range, where it occurs in relatively low numbers, is also important to increase the chances of genetic exchange between populations. The predictions of relatively high-abundance populations in the north of the beetle's range were not borne out by the field-checking. Examination of the spatial distribution of the residuals (Fig. 1) indicated that the model was over-estimating the species abundance in this area. This information can not only be used to inform the development of a conservation management strategy for the species, but also indicate where further sampling should be focussed in any attempt to improve the predictive ability of the model.

Conservation management considerations

In order for the results of this study to be of use to land managers a positive correlation between habitat quality and abundance of *H. simsoni* needs to be assumed. Van Horne (1983) and Elith (2000) caution against such an assumption as individuals of a species may not congregate in the most suitable locations because of behaviour, competitive interactions, dispersal dynamics and historical factors such as disturbance history. For invertebrates in particular, it is well recognised that species are often patchily distributed and do not occupy all apparently suitable habitats within their range (Hill & Michaelis 1988). However, as pointed out by Pearce & Ferrier (2001), obtaining detailed demographic and resource utilisation information is prohibitively intensive and expensive for many species and may delay conservation planning activities that require immediate action. For *H. simsoni*, such detailed information is not available across large parts of its range, hence variation in species abundance is used as the best predictor available for assessing habitat quality. Therefore, we have assumed that the predicted categories of abundance and probability of presence of

H. simsoni represent predictions of optimal, sub-optimal and marginal habitat for the species, with the "High" category equivalent to optimal habitat.

Whilst potential wet forest habitat makes up 73% of the 250 km^2 range of *H. simsoni*, very little of this constitutes optimal and sub-optimal habitat for the species. The majority of the species' range consists of marginal habitat (Table 3). Predicted optimal habitat makes up only 2,100 ha (or 12%) of the potential habitat available to the species, with sub-optimal habitat making up a further 30% (Table 3). In terms of protection of predicted optimal and sub-optimal habitat, populations of *H. simsoni* are poorly reserved across its range. Only 400 ha of predicted optimal habitat occurs within Tasmania's Comprehensive, Adequate and Representative (CAR) reserve system. Twenty-four percent of sub-optimal habitat occurs in CAR reserves. Prior to this study over 52% of optimal habitat and 45% of sub-optimal habitat on public land were likely to be subject to forestry activities (Table 3). A further 10% and 16% respectively occur on private land and hence are of uncertain future (Table 3). There is a high correlation between the distribution of quality habitat for *H. simsoni* and forest identified with a moderate-high potential for plantation establishment. Ignoring land tenure, 91% of optimal habitat and 68% of sub-optimal habitat have at least a moderate potential for plantation. These figures are probably over-estimates, but demonstrate the need for the development of 'off-reserve' conservation management strategies for *H. simsoni*.

Table 3. The distribution of the three categories of predicted frequency of occurrence of *H. simsoni* in wet eucalypt forest and mixed/rainforest according to land-use (prior to any species-specific land-use changes on State forest). Other public land category includes miscellaneous land uses such as Hydro Electric Commission land. All figures are in hectares (ha).

Land management category	High (0.5 - 1.0)	Medium (0.2 - <0.5)	Low (< 0.2)	Total predicted habitat
Formal reserves	100	800	3 800	4 700
Informal reserves	300	500	800	1 600
Other public land	0	100	100	200
State forest (couped)	1 100	2 500	3 600	7 200
State forest (uncouped)	400	900	1 000	2 300
Private Property	200	700	1 300	2 200
Total	2 100	5 500	10 600	18 200

Source: GIS Section, Forestry Tasmania.

The intensification of forestry activities in 'off-reserve' forested areas recently increased following Tasmania's *Regional Forest Agreement* (Commonwealth of

Australia & State of Tasmania 1997), resulting in an upsurge in clearing of the native forests in the north-east of the State for conversion to plantations (Munks & McArthur 2001; Lindenmayer & Franklin 2002). In Chapter 1 it was found that the conversion of potential wet forest habitat to plantation results in the local extinction of *H. simsoni*. They also suggested that whilst clearfell, burn and sow regimes are likely to have a significant negative impact on populations of the beetle, recovery of beetle populations may occur within a standard 80-90 year harvest rotation, with recolonisation of disturbed habitat through immigration from adjacent undisturbed forest. This information on potential threats and conservation requirements (Chapter 1) can be coupled with the map of predicted habitat derived from the frequency of occurrence model to inform the development of a conservation management strategy for *H. simsoni*.

Any strategy developed for *H. simsoni* should specifically conserve or reduce threats in areas predicted by the model to contain medium and high quality habitat. A comparison of observed frequencies of occurrence with corresponding densities of *H. simsoni* showed that areas predicted as high quality habitat contain exponentially greater populations of beetles than areas of lesser quality (Fig. 4). In fact, populations in high quality habitat potentially make up half of the estimated total population of the species (Table 4). The importance of these areas of optimal habitat to the species has been reinforced by recent work modelling the population viability of the species, based on the frequency of occurrence habitat model developed in this study, under various management scenarios (Fox, Meggs, Munks, McCarthy & Burgman unpublished data). The results of the PVA indicate that if there is ongoing protection of contiguous areas of optimal habitat, the risk of extinction of the species is negligible. Therefore, it is recommended that areas of optimal habitat be incorporated within Tasmania's CAR reserve system to ensure their ongoing protection in accordance with Tasmania's *Regional Forest Agreement* (Commonwealth of Australia & State of Tasmania 1997).

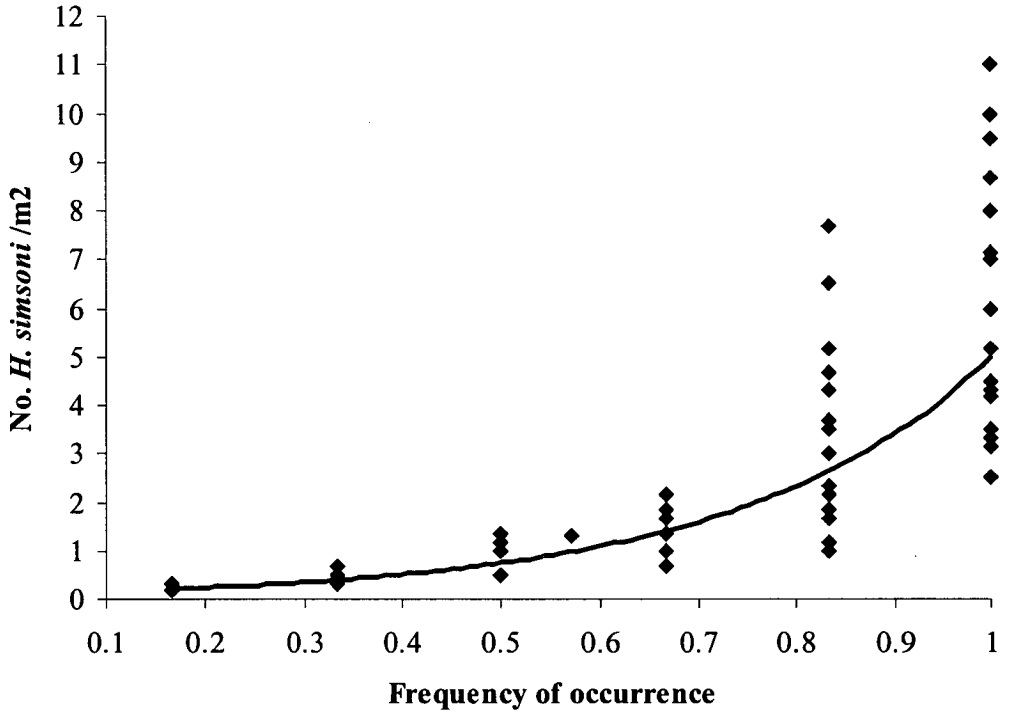

Figure 4. Relationship between the frequency of occurrence and density of *H. simsoni* at the evaluation sites (n = 88).

Table 4. Estimated total population size of *H. simsoni* based on the relationship between the frequency of occurrence and the number of beetles per hectare (n = 88) multiplied by the quantity of each category of habitat quality. These figures are likely to be an over-estimate as they are based on the assumption that every patch of predicted habitat is occupied.

Frequency of occurrence. (habitat quality)	**No./m^2** (mean ± se)	**#Adj No./m^2** (mean ± se)	**Est No./ha** (mean ± se)	*Total no. of beetles*
HIGH (Optimal)	3.87 ± 0.39	1.29 ± 0.13	12,900 ± 1,300	27,090,000 ± 2,730,000
MED (Sub-optimal)	0.56 ± 0.07	0.19 ± 0.02	1,900 ± 200	10,450,000 ± 1,100,000
LOW (Marginal)	0.22 ± 0.02	0.07 ± 0.01	700 ± 70	7,420,000 ± 742,000
Estimated total species population size				44,960,000 ± 4,572,000

#Adjustment assumes that hand-collecting of dead beetles samples at least three beetle population cycles.

Conclusions

To our knowledge this is the first study that reports on not only the development of predictive distribution and abundance models for a threatened invertebrate but also on their statistical strength and usefulness for conservation purposes. It demonstrates a way in which the distribution of habitats important to a forest-dependent threatened invertebrate can be predicted at a scale appropriate to forest management. Land managers need to have confidence in such predictions since inappropriate land-use decisions in the conservation management of threatened species can lead to potential species extinction and/or unnecessary economic and social impacts resulting from well-intentioned, but misguided land-use changes. An evaluation dataset independent of the model was used in a rigorous test of the model's predictive ability as recommended in other studies. This study has also demonstrated that model accuracy and model utility are not the same thing. It is important to remember the original goals of a study in choosing model evaluation methods. Hence, a range of criteria were used to assess the models' predictive ability. In this study, the aim was to identify the extent and spatial arrangement of habitats important to this species and the areas in which conservation management of the species may conflict with planned forestry activities. Statistically, the frequency of occurrence model performed best as a simple presence/absence predictor, but as such has much less utility for conservation management than as a predictor of abundance (or habitat quality). It was clear from the original sampling (Chapter 1), that there were definite "hotspots" of abundance - areas where there was a magnitude greater abundance of *H. simsoni*, and the evaluation analysis has shown that the model performs best in predicting the occurrence of such areas. It is for this reason that we conclude that whilst the frequency of occurrence model had only exhibited a moderate level of statistical strength, it has a high level of utility as a conservation planning tool for *H. simsoni*. The usefulness of a model in planning the conservation of a species and its habitat should not necessarily be discounted if the model is found to be inaccurate according to some criteria. In the absence of information on many aspects of the biology and ecology of *H. simsoni*, it is proposed that land managers can confidently utilise the predictive habitat map derived from the frequency of occurrence model in this study to inform the development of conservation measures throughout the species' range.

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CHAPTER 3

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DISTRIBUTION, HABITAT CHARACTERISTICS AND CONSERVATION REQUIREMENTS OF A FOREST-DEPENDENT THREATENED INVERTEBRATE *LISSOTES LATIDENS* (COLEOPTERA: LUCANIDAE)

*Jeff M. Meggs[1,2] and Sarah A. Munks[1]

[1]Forest Practices Board, 30 Patrick Street, Hobart, Tasmania.

[2]Present address, Forestry Tasmania, 79 Melville Street, Hobart, Tasmania.

*To whom correspondence should be addressed: jeffrey.meggs@forestrytas.com.au

ABSTRACT

The distribution, habitat and conservation requirements of an endangered stag beetle, *Lissotes latidens* (Coleoptera: Lucanidae), in south-east Tasmania were investigated. This study increased the number of known localities of *L. latidens* from 10 to 34 and almost trebled the known range of the species to 280 km^2. The beetle was rediscovered at its type locality on Maria Island, where its range is in the order of 5 km^2. Although genetic algorithm modelling predicted suitable environmental conditions for the species on the Forestier and Tasman Peninsulas, subsequent surveying failed to find the species in these areas. Relative to other Tasmanian lucanids with similar habits, *L. latidens* was found to occur at quite low population densities across its range. The beetle was found to inhabit a range of wet forest types including damp eucalypt forest, wet eucalypt forest, rainforest and wet/damp eucalypt forest patches and riparian areas amongst drier forest types. Dry eucalypt forest was found to be unsuitable habitat. Within its range, 42 km^2 of forest was identified as potential habitat, representing 15% of the species' total range. Extensive areas of wet forest habitat were concentrated in the eastern half of its range and on the northern part of Maria Island. The relative isolation and small size of wet forest patches and riparian areas in the western half of the species' range brings into question the long-term viability of populations of *L. latidens* in these areas. *L. latidens* had a preference for forest with a well-developed overstorey and greater than 10% ground cover of coarse woody debris (CWD). This study also revealed that although the species is soil-dwelling throughout its life-cycle, it has a close association with CWD, occurring under logs at the interface of soil and CWD. With the exception of the wet forest on Maria Island National Park, suitable habitat for *L. latidens* is poorly reserved. Fourteen percent of the potentially suitable habitat for the species is privately owned forest, whilst 48% is managed as wood production forest by Forestry Tasmania under clearfell, burn and sow silvicultural regimes; a practice that is likely to lead to the depletion of CWD over successive harvesting rotations. Recommendations are made for the conservation management of habitat utilised by *L. latidens* in 'off-reserve' areas subject to production forestry. The need for conservation strategies to incorporate the temporal dynamics of habitat important to forest-dependent threatened species is also discussed.

Keywords: invertebrate conservation; coarse woody debris; riparian forest; forestry; lucanid beetles; 'off-reserve' conservation; threatened species.

INTRODUCTION

Much of the focus on the conservation of forest-dependent biodiversity in Tasmania has been directed at the expansion of existing protected areas to achieve a Comprehensive, Adequate and Representative (CAR) reserve system. The most recent example of this was the addition of 400,000 ha of new reserves established under the *Regional Forest Agreement* (RFA) process (Commonwealth of Australia and State of Tasmania 1997). This has resulted in a more representative reserve system in terms of forest communities and the protection of 40% of the State's forest (RPDC 2002). The success of this strategy in the conservation of forest biodiversity, the bulk of which comprises invertebrates, largely depends on whether there is a high correlation between the spatial arrangement of the reserve system and the distributions of a significant proportion of Tasmania's invertebrate fauna.

A large proportion of habitats important for threatened species, particularly invertebrates, remain in the 'off-reserve' landscape (Munks *et al.* submitted). Seventy-nine species of forest-dependent invertebrates with at least part of their range occurring in production forest were listed in the original schedules of the Tasmanian *Threatened Species Protection Act* 1995 (Jackson and Taylor 1995). Despite the significant improvements to the reserve system resulting from the RFA, only one of these species, *Anoglypta launcestonensis* (an endemic land snail), has been delisted since the proclamation of the Act, whilst a further 21 species have been added (Forest Practices Board 2002; RPDC 2002). This is in part due to the obvious limitations of any reserve system capturing adequate levels of habitat for all species. However, it may also be a consequence of the increase in intensive forest management practices in 'off-reserve' areas that has been concomitant with the expansion of the reserve system (Forestry Tasmania 1998a; Lindenmayer and Franklin 2002). Whilst habitats important to threatened forest-dependent invertebrates have been captured within the

CAR reserve system, there is a clear need to combine reservation with 'off-reserve' conservation measures to ensure the long-term viability of many species.

Critical to the development of a conservation strategy for any threatened species is an understanding of the distribution and habitat characteristics of the species, as well as the specific threats to the species from land-use practices. For many of the invertebrates listed in the schedules of the Tasmanian *Threatened Species Protection Act* 1995 comprehensive distribution and habitat utilisation data are not available (Taylor and Bryant 1997; Munks and Taylor 2000). The broad-toothed stag beetle, *Lissotes latidens* (Coleoptera: Lucanidae) is an endemic Tasmanian beetle and is listed as Endangered under the Tasmanian *Threatened Species Protection Act* 1995 due to its restricted distribution, low population density and loss of habitat (Forest Practices Board 2001). It is a medium-sized, flightless black beetle, with a body length varying between 12 and 18 mm. The males have prominent "bulls-horn" shaped mandibles. There have been no detailed surveys of the distribution or reservation status of the species, however it has been thought to be extremely localised and to occur at low population densities (K. Michaels pers. comm.). Prior to this study it had been recorded at ten locations in eastern Tasmania. Seven of these were from within Wielangta State Forest (Michaels 1996 and 1999; Blake unpubl. data; G. Bornemissza pers. comm.). Two records were from west of this area, at Adams Hill (Munks unpubl. data.) and Bust-me-gall Hill (G. Bornemissza pers. comm.). Hence, the range of the species on mainland Tasmania was estimated to be in the vicinity of only 90-100 km^2 (Fig. 1). The type locality for *L. latidens* is Maria Island (Westwood 1855), but the beetle was last collected there in the 1960's (P. McQuillan pers. comm.). These species' records occur in a continuum of forest communities, from dry eucalypt forest through to rainforest (Michaels 1996). Anecdotal information suggested that decaying logs and/or stumps are very important for all stages of the lifecycle. However, there was some question as to whether the species is log-dwelling but active outside logs for some of the year (or for some stages of its life-cycle), as with other lucanid species, or edaphic (soil-dwelling), requiring logs for shelter.

The major threatening process to the survival of *L. latidens* is loss of its forest habitat (Michaels and Bornemissza 1999). Studies of related species have indicated that intensive forest management activities can have a severe negative impact on soil and

log-dwelling lucanids (Chapter 1; Chapter 4). The aim of this study was to obtain further information on the distribution and habitat requirements of *L. latidens* vital for the formulation of any conservation management initiatives required to ensure its security in areas subject to production forestry.

METHODS

Study area

The study was conducted between November 1997 and December 1998 in south-east Tasmania. The study area incorporated the known localities of *Lissotes latidens* on mainland Tasmania and all of Maria Island (Fig. 1). Sites on private land, State forest, and reserves managed by Forestry Tasmania or the Tasmanian Parks and Wildlife Service were sampled.

The forest types within the study area vary widely and can change with aspect within relatively short distances from dry to wet eucalypt forest (Forestry Tasmania 1998b). Patches of relict rainforest also occur in humid fire-shadow environments such as deep gullies and are generally dominated by *Atherosperma moschatum* (sassafras) (Neyland 1991). Wet eucalypt forests in the study area have a rainforest or wet sclerophyll understorey, and tend to occur in relatively humid or shaded environments. *Eucalyptus obliqua* communities dominate the wet forest in the area, although remnant stands of *E. regnans* also occur (Forestry Tasmania 1998b). Dry eucalypt forests and woodlands in the study area have understoreys dominated by narrow-leaved shrubs, grasses or graminoids, and occupy drier, more exposed or infertile sites. Some dry forest communities contain several co-occurring eucalypts including *E. amygdalina*, *E. pulchella*, *E. tenuiramis*, *E. obliqua* and *E. globulus*. Wet and dry eucalypt forest types are often intermixed in broad transition zones throughout the study area. The damp forest type recognised in this study fell within these transition zones. Mean annual rainfall within the study area is 850 mm, but varies between 600 mm and 1000 mm (Forestry Tasmania 1998b). The dominant rock-type is Jurassic dolerite. The majority of the study area was private land, but also included areas of State forest managed by Forestry Tasmania and Formal Reserves managed by the Department of Primary Industries, Environment and Water (DPIWE).

The study area is of particular ecological and evolutionary interest. It incorporates the Wielangta Indicative Area that was identified in the *Regional Forest Agreement* as having National Estate value as an area of high fauna species richness (PLUC 1997). It was recommended that this area required protection beyond that provided by current formal and informal reserves and pre-existing forest management systems. Although there have been few studies of the invertebrate fauna of the study area, relative to the rest of Tasmania it is one of the better-sampled areas (Mesibov 1996).

Site selection

Following a preliminary survey of known localities of *L. latidens* to determine the broad characteristics of habitat in which the species occurred, four broad forest types were recognised for the stratification of the survey. These included wet eucalypt forest (24 sites), damp eucalypt forest (8), dry eucalypt forest (10) and riparian wet forest (15). The latter forest type encompassed rainforest, damp eucalypt forest and wet eucalypt forest corridors and patches within a matrix of dry eucalypt forest. Areas of similar vegetation communities were identified using 1:25000 Photo Interpretation (P.I.) maps produced by Forestry Tasmania (Stone 1998). These maps identify the height and density of the tree layer, with eucalypts being distinguished from rainforest trees and acacias, provide some information of the understorey, and code the area accordingly. P.I. maps do not distinguish between wet and dry eucalypt forest. Wet eucalypt forest was assumed to be represented by PI codes of E+3 and above, E-3 and below if in conjunction with T(W), and T(W) on its own. Dry eucalypt forest was assumed to have a PI height potential code of E-3 and below that was not accompanied by codes indicating wet forest (eg. T, M, T(W), Mr). Information on the likely occurrence of damp eucalypt forest and wet riparian forest in areas coded as dry forest was obtained from Forestry Tasmania Derwent District staff and North Forest Products Triabunna staff. Additional criteria used to select sampling sites included: accessibility (for logistic reasons generally < 500 m from roads); occurrence within or immediately outside (ie, < 5 km) the known distribution of the species; and proximity to other sites, with the aim of maximising the geographic range sampled within the study area.

Animal Survey

Eight of the known localities of *L. latidens* on mainland Tasmania were sampled initially to determine the optimal animal sampling technique. At each known locality the following sampling methods were tested within a 25 m radius plot:

(i) the underneath of all logs in the plot that could be moved by hand was searched for adult *L. latidens* and larval lucanids;

(ii) a sub-sample of these logs was broken open and rotting wood was searched for adult and larval stag beetles (as per Chapter 4);

(iii) in a sub-plot of 10 m radius, six 1 m^2 leaf litter plots were searched for live adult *L. latidens* or body parts of dead ones (as per Chapter 1).

Pitfall traps were not used to sample the species for ethical reasons and because hand-collecting of litter invertebrates has been found to be more time-efficient and to provide more accurate data on species abundance than pitfall trapping (Mesibov *et al.* 1995). In addition, the previously-recorded low level of trapping success for this species using pitfalls (Blake unpubl. data; Michaels 1996 and 1999) suggested that this method had limited potential for sampling populations of this beetle.

Rolling logs and searching under them proved to be the only reliable method for locating *L. latidens*. Therefore this method was used for the remainder of the study. At each site, the underneath of all logs that could be moved by hand within a 25 m radius plot was searched for adult *L. latidens* and larval lucanids. The diameter of logs that could be sampled was thus generally less than 50 cm, however short lengths of log with larger diameters were sampled wherever possible. Any live beetles were recorded and released at the site of capture. Parts of dead beetles were recorded and lodged with the Forestry Tasmania Insect Collection, Hobart and the Queen Victoria Museum and Art Gallery, Launceston. Identifiable body parts included male heads and abdomens of both sexes, which have a distinctive pattern of ridges relative to other lucanid species (G. Bornemissza pers. comm.). The measure of abundance of *L. latidens* used in this study was the number of live beetles plus the minimum number of dead individuals (as indicated by the number of body fragments) found in each 25 m radius plot (as per Chapter 1).

In an attempt to establish the larval habit of the species both log-dwelling and soil-dwelling lucanid larva were collected with a small quantity of the substrate in which they were found at eight of the study sites where *L. latidens* was known to occur. The larvae were reared in the dark at laboratory temperature (10-22°C). The larvae were checked every seven to 14 days for parasites, at which time the moisture level of the substrate was also checked. Specimens that were successfully reared to adulthood were identified to species level using Lea (1910).

Habitat variables

Habitat variables recorded at each site were chosen for their anticipated value as predictors of beetle distribution and abundance, and for the ease with which they could be collected (as per Chapter 1). The following habitat variables were assessed at each site (ie, each 25 m plot): altitude (m), distance to nearest stream (<30 m; 30-100 m; >100 m), leaf litter depth (<1 cm; 1-3 cm; >3 cm), leaf litter cover (% ground cover), rock cover (very low; low; medium; high), geology (from 1:50000 Geological atlas maps), CWD cover (% ground cover of logs >10 cm mid-diameter), CWD diameter-distribution (1 = < 10cm, 2 = 10-50cm, 3 = > 50cm, 4 = all three, 5 = mix of two (specified)), moss cover (% ground cover including on rocks and logs), average aspect (N, S, E, W, none), average slope (degrees), and topographic position (1 = flat, 2 = gully, 3 = slope, 4 = ridge).

The species composition and structure of the forest community was recorded at each site (as per Chapter 1). Floristic nomenclature was as per Buchanan (1999). Forest structure was measured by categorising the forest at each site into vegetation height classes including: overstorey tree, understorey tree, tall shrub, low shrub and ground cover. For each height class present at a site the average height (m) and average canopy cover (%) was visually estimated.

A description of the type and level of disturbance at each site was recorded, taking into account such factors as: selective logging (none, low, medium or high), fire (recent (ie, <50 years ago), or not recent), and distance to nearest road (m). Soil profile descriptions were also made at each site following the format of Grant *et al.* (1995). Where an adult *L. latidens* or larval lucanid was found under a log, notes were made of

the tree genus, diameter at the mid-point of the log, decay class of the log (as per Meggs 1996) and a description of the extent and depth of contact between the log and the soil surface.

Statistical analyses

Generalised Linear Modelling

The relationship between the abundance of *L. latidens* and the habitat variables recorded for each site was analysed using Generalised Linear Models (GLM) (McCullagh and Nelder 1989). Dry eucalypt forest was excluded from all analyses because preliminary results indicated that it does not constitute suitable habitat for this species. The model was fitted assuming a Poisson distribution and log-link function. A stepwise-backward regression process was used in which variables were retained if their removal from a full model resulted in a sufficiently large change in deviance. A large deviance was chosen so that all the variables that were finally in the model were highly significant. This has the effect of simplifying the model. Since more variables could have been added, the resulting model should be considered conservative. No interactions were fitted, further simplifying the model. Some habitat variables needed logarithm or square-root transformation to decrease excessive skewness. The relationship between the presence/absence of *L. latidens* and the habitat variables recorded for each site was also modelled (assuming a binomial distribution and logit-link function). Levels of statistical significance were set at 0.001 because of the relatively small size of the dataset and because no interactions were fitted.

Genetic Algorithm Modelling

The environmental modelling package CORTEX (Peters and Thackway 1998) was used to predict the potential distribution of *L. latidens*. CORTEX is a form of modelling that combines artificial intelligence techniques with the BIOCLIM (Nix 1986) approach (Peters and Thackway 1998). CORTEX has been tailored specifically for Tasmania and uses environmental variables calculated from the 200 metre grid cell Tasmanian GIS surfaces. Thirteen environmental variables are used: three elevation models, a relief model, a slope model, an aspect model, topographic wetness index, annual mean temperature, annual mean rainfall, annual temperature range, coefficient of variation of mean monthly rainfall, soil fertility index, and geology

(Brereton 1997). Means and modes of these variables are used at a scale of 1 km grid squares. An environmental envelope model is constructed using the rule-based genetic algorithm procedure and a map of the predicted distribution of the species is produced at the 1 km grid square scale (Peters and Thackway 1998).

RESULTS

Distribution and relative abundance of Lissotes latidens

Lissotes latidens was present in 25 of the 58 sites sampled and has now been recorded from 34 localities in south-east Tasmania, 28 on mainland Tasmania and six on Maria Island (Fig. 1). The range of the species was extended in the present study in most directions except to the west. The recalculated extent of occurrence of *L. latidens*, calculated as the area contained within the shortest boundary drawn that encompasses all known sites of occurrence (IUCN Species Survival Commission 1994) is now 280 km^2 (27,950 ha). This has almost trebled the previous known distribution of 98 km^2. The majority of the beetle's range occurs on mainland Tasmania (274 km^2), whilst its range on Maria Island is 5.4 km^2.

Lissotes latidens was generally found to occur at low densities (0.0005-0.003/m^2). Only 32 of the 53 beetles found in this study were alive, so densities may over-estimate the size of extant populations because body parts may accumulate over a number of reproductive cycles, which are believed to be annual (G. Bornemissza pers. comm.).

The CORTEX analysis of the environmental domain of *L. latidens* produced predicted extensions of the range of the species mainly to the north and south (Fig. 2). This model predicted a moderate probability of the species' occurrence south of the study area, on the Forestier Peninsula and in the western half of the Tasman Peninsula. Subsequent surveys for the species in potential habitat for *L. latidens* on the two peninsulas failed to find the species.

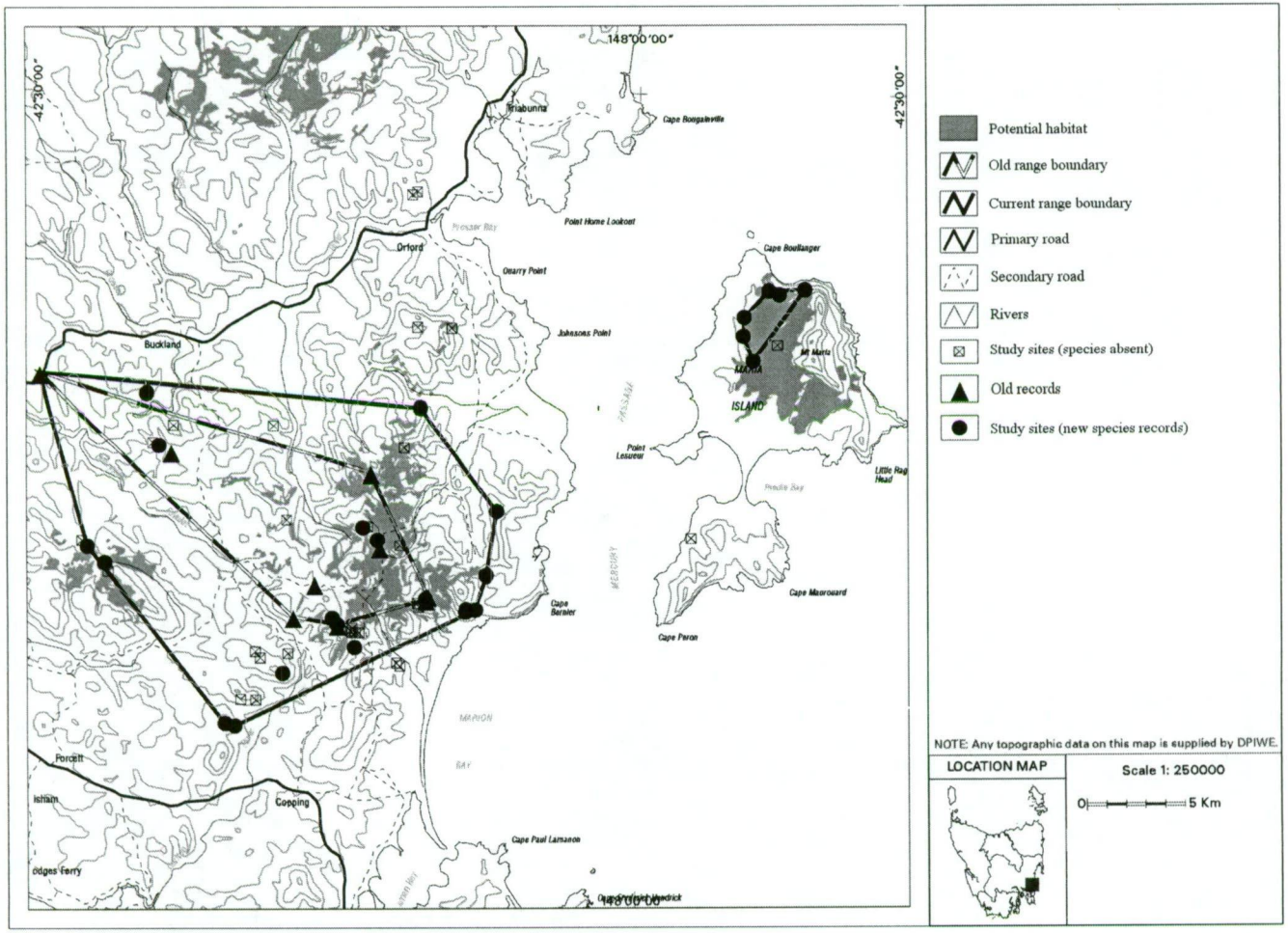

Figure 1. Distribution of *L. latidens* and potential habitat for the species on mainland Tasmania and Maria Island. The sites sampled in this study, including sites of species' absence, and old species records are shown. The type locality of the species was recorded as 'Maria Island' and hence is not illustrated as an 'old record'.

Relationships between the occurrence of* L. latidens *and habitat variables

Vegetation was found to be the only reliable predictor of the presence/absence of *Lissotes latidens*. The species was not found in dry forest communities, however it was found to occur in the transition zone between wet and dry eucalypt forest (8 sites) and in patches of wet/damp forest along drainage lines and in riparian corridors surrounded by dry forest (10 sites).

In general, the forest habitat in which *L. latidens* was found can be summarised as wet eucalypt forest dominated by *Eucalyptus obliqua*, *E. regnans* and *E. globulus*, with occasional *E. viminalis*, and wet/damp eucalypt forest patches and riparian areas amongst drier forest types. The damp eucalypt forest patches/riparian areas were typically dominated by *E. obliqua* but with *E. viminalis*, *E. pulchella*, *E. globulus* and *E. tenuiramis* often present. The wetter patches/riparian areas were generally dominated by *E. obliqua* or *E. globulus*. Understorey vegetation ranged from typical broad-leaved wet sclerophyll plants such as *Pomaderris apetala* and *Olearia argophylla* in the wetter forest/riparian areas to a mixture of broad-leaved and narrow-leaved plants such as *Zieria arborescens*, *Cyathodes glauca*, *Pultenaea juniperina*, *Acacia verticillata* and *Lomatia tinctora* in the damp forest/riparian areas.

There did not appear to be any relationship between soil physical characteristics and the occurrence of *L. latidens*. The soils at the majority of sites surveyed in the study area fit the descriptions of Grant *et al.* (1995) for soils on Jurassic dolerite under dry and wet forest. Topsoil textures ranged from alluvial sands through to heavy clays, but could generally be described as loams, clay loams and light clays.

Numbers of beetles did not differ significantly between forest types. The GLM analysis of beetle habitat indicated that the abundance of *L. latidens* was dependent on the following variables (Table 1):

No. beetles = Constant + CWD cover + Forest type + Low shrub canopy cover + Leaf litter cover + Overstorey tree canopy cover + Understorey tree canopy cover + CWD diameter distribution

Table 1. Estimates of regression coefficients for the GLM of the relationship between numbers of *L. latidens* and habitat variables recorded at each site (s.e. = standard error; t = the t-statistic, used as a rough guide to test whether each of the factor levels differ from the first level; * = variables that were significant at p<0.001).

Habitat variable	Estimate	s.e.	t(*)	p
Constant	-4.740	1.590	-2.98	0.999
CWD cover (6-10%)	0.701	0.600	1.17	0.121
CWD cover (>10%)*	2.663	0.716	3.72	<0.001
Forest-type (damp)	1.411	0.554	2.55	0.005
Forest-type (riparian)	0.925	0.400	2.31	0.010
(Log) Low shrub canopy cover	-1.306	0.573	-2.28	0.989
(Log) Forest cover (50-75%)	1.030	0.507	2.03	0.021
(Log) Forest cover (>75%)	-0.244	0.382	-0.64	0.738
Overstorey tree canopy cover*	0.116	0.033	3.50	<0.001
(Sqr) Understorey tree canopy cover	0.497	0.182	2.73	0.003
CWD diam dist (all sizes)	-0.500	0.673	-0.74	0.771
CWD diam dist (small & med.)*	1.964	0.564	3.48	<0.001
CWD diam dist (med. & large)	0.977	0.402	2.43	0.008

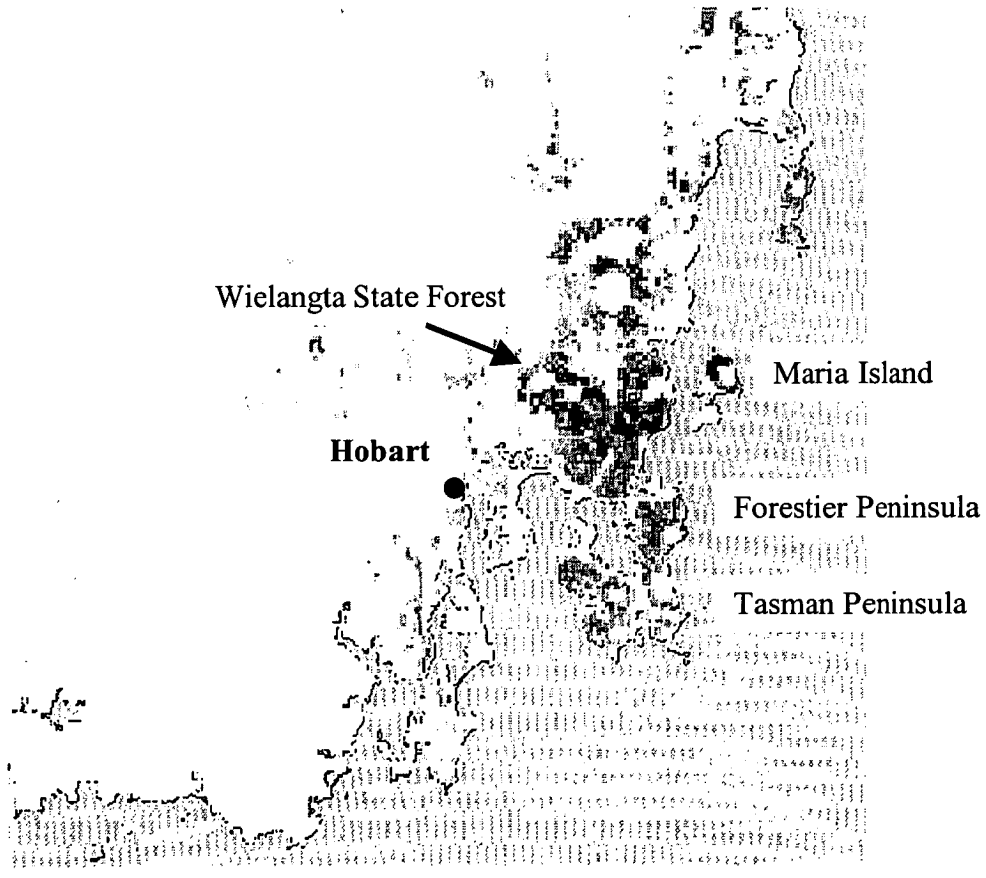

Figure 2. Predicted distribution of *L. latidens* from the CORTEX model. The darker the shading, the higher the probability of presence.

Significantly higher numbers of *L. latidens* were found at sites with greater than 10% ground cover of CWD (Fig. 3). In addition, significantly more beetles were found at sites where small (<10 cm) and medium (10-50 cm) diameter logs were the dominant size classes of CWD (Fig. 4). There was also some indication that the occurrence of medium and large (>50 cm) logs was influencing the abundance of *L. latidens* (Table 1).

Forest structure was found to have a significant influence on the number of *L. latidens* found at each site. Numbers of beetles increased as the percentage canopy cover of the overstorey tree layer increased (Fig. 5). There was also a trend for beetle numbers to increase with increasing canopy cover of the understorey tree layer, although this was not statistically significant (Table 1).

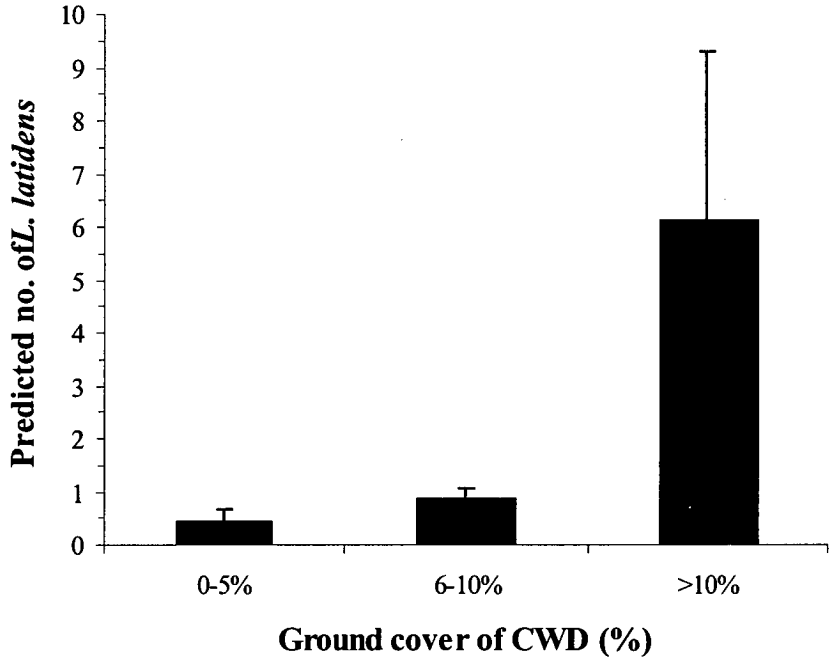

Figure 3. Relationship between the predicted abundance of *L. latidens* and the amount of ground-cover of coarse woody debris (CWD).

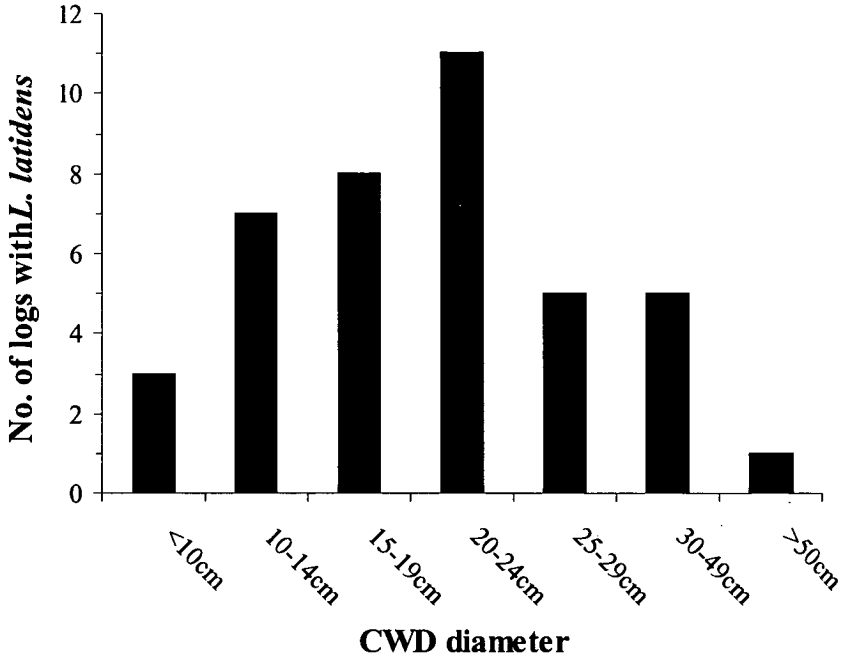

Figure 4. Relationship between the occurrence of *L. latidens* and the diameter size-class of CWD.

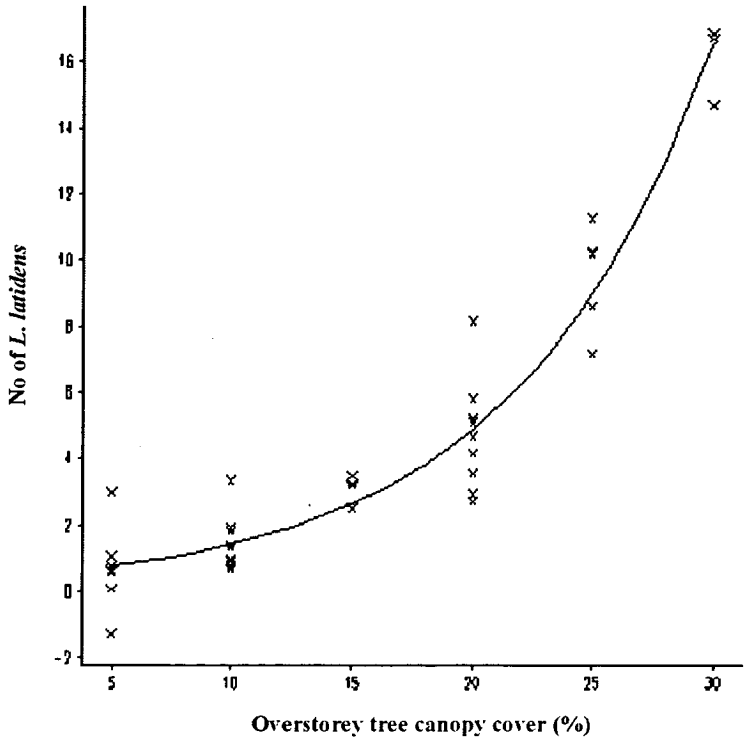

Figure 5. Relationship between predicted and observed abundances of *L. latidens* and the percentage canopy cover of the overstorey tree layer.

Microhabitat characteristics

Of the 54 adult *Lissotes latidens* found in this study, 46 were located under fallen dead wood, five were observed wandering on top of leaf litter or roads, and three were identified from fragments found amongst leaf litter.

Seventy-five percent of the logs under which *L. latidens* was found were identified as eucalypt logs, although it was also found under musk, dogwood and blanket bush logs. All but one of these logs was in the early to moderate stages of decay (Fig. 6). All of the adult *L. latidens* found under logs were located either on the soil surface at the soil/log interface or were partly buried within the upper soil layer. Seven beetles were observed burying themselves into the soil when released at the point of capture.

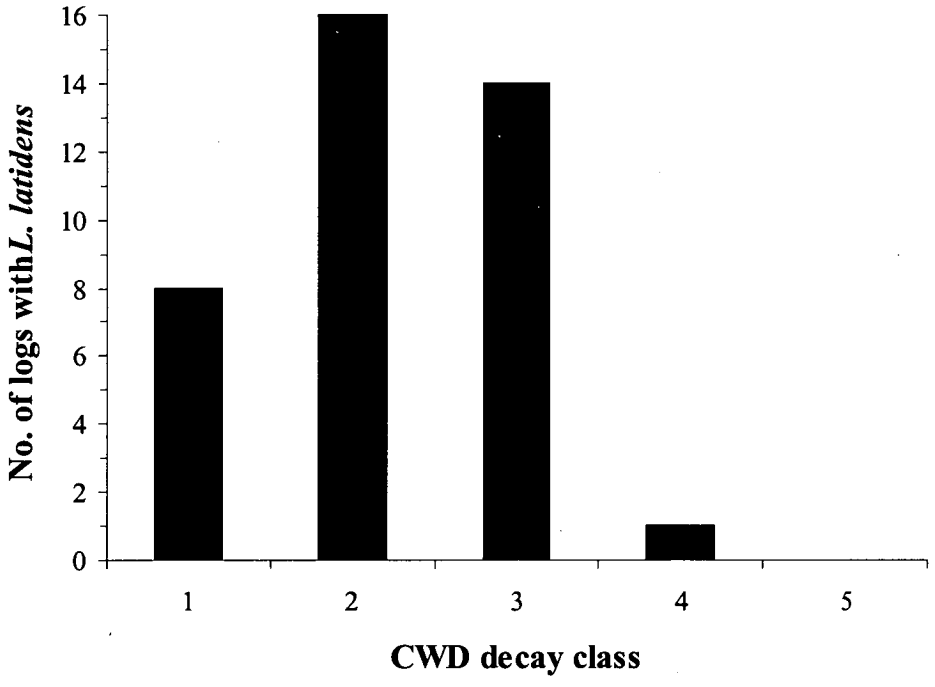

Figure 6. Relationship between the occurrence of *L. latidens* and CWD decay class (1 = least decayed, as per Meggs 1996).

Lissotes latidens appears to be soil-dwelling during the larval stage of its life-cycle. Five larvae collected from shallow depressions in the soil at the soil/log interface survived to adulthood. One of these larva developed into an adult male *L. latidens*. All log-dwelling and all other soil-dwelling larvae developed into *Lissotes obtusatus*, a common stag beetle in southern Tasmania.

Occurrence of Lissotes latidens *and habitat disturbance*

L. latidens was found at sites showing evidence of variable levels of disturbance. Just over half of the sites at which it was found had been burnt by wildfire or prescribed burning in the last 50 years. At three of these sites the species was found under logs that had been severely scorched in these fires.

Fifty-nine percent of recorded localities of *L. latidens* showed evidence of selective logging activity, with harvesting levels varying from a few stems to many stems per hectare. Two records of the species occurred in forest regenerating from clearfell, burn and sow (CBS) forestry operations in the 1970's.

L. latidens was found to occur in the streamside reserves (ie, riparian buffers) retained in two eucalypt plantations. It was not found to occur in windrows within the plantations, nor were any lucanid larvae observed in these windrows.

DISCUSSION

Distribution and relative abundance of Lissotes latidens

Although the known range of *Lissotes latidens* on mainland Tasmania has been almost trebled to 275 km^2, it is unlikely that future searches will extend this range in a contiguous fashion any more than a few kilometres beyond that presented here. In terms of broad forest types, unsuitable habitat surrounds much of the beetle's range (Fig. 1). However, on Maria Island potential habitat extends well beyond the current known range of the species. Hence, further surveying of the island could increase its range from 5.4 km^2 to greater than 20 km^2 (Fig. 1).

L. latidens appears to occur at relatively low population densities. Generally only one or two specimens were located at a site. The highest number of beetles found at a site was six (two sites), which is equivalent to a density of 0.003/m^2 (30 beetles/ha). This compares with densities of up to 10/m^2 (100,000 beetles/ha) for *Hoplogonus simsoni* in north-east Tasmania, a threatened lucanid with an equivalent sized range, patchy distribution, and similar larval and adult habits (Chapter 1). It should be noted,

however, that the sampling method used in the present study is based on a number of assumptions and has some limitations in the determination of the species' presence and hence also the estimation of absolute population sizes. The most important assumption was that there is a close association between the species and fallen-dead wood and that this is consistent across time and space. It is possible that this microhabitat is only one of a number used by *L. latidens*. Specimens have been found amongst leaf-litter and one (a dead specimen) has even been collected from inside a rotting log (Michaels 1996). It is possible that sites with a relatively low quantity of CWD at which the species was not found, have been mistakenly ascribed as sites of species' absence or low-density when alternative micro-habitats were not investigated. It is also possible that in drier parts of the species' range, such as in damp forest, fallen-dead wood is the preferred microhabitat due to its relative insulation from extremes of temperature and water loss. Hence, in these areas populations sizes may have been over-estimated. Conversely, the limited application of the log-rolling method for sampling under large diameter logs may mean that densities have been underestimated, given the relatively greater surface area of potential microhabitat they provide. Clearly, the strength of the association between *L. latidens* and CWD requires further investigation.

Habitat characteristics

At a regional level, the restricted distribution of *L. latidens* appears to be partly due to an interrelationship between climate and topography, and hence rainfall. The CORTEX model indicated that the species occurs in an area of relatively high rainfall for this part of Tasmania. The variability in rainfall across this region is reflected in the changing distribution of wet sclerophyll forest throughout the area. Wet forest is widespread in those parts that have a relatively high rainfall (eg. the Wielangta area, the Tasman and Forestier Peninsulas and northern Maria Island), in relatively cool and moist upland environments (eg. south and east facing slopes on the outskirts of the Wielangta area), and is restricted to riparian areas in the driest parts of the region (eg. west of Wielangta) (Duncan 1991). The species' absence from the Forestier and Tasman Peninsulas indicates that the occurrence of suitable rainfall and vegetation types is not sufficient to explain the current distribution of this species. Neither can geographic isolation explain this phenomenon given the occurrence of the species on Maria Island which has been separated from mainland Tasmania for the last 10,000

years (Dept. Environment and Planning and Dept. Parks, Wildlife and Heritage 1993). It is unclear whether the absence of the species from these areas is due to habitat characteristics other than those recorded in this study or whether it may be due to other factors, such as interspecific interactions or historical events.

Only broad forest type proved to be a good predictor of the likely presence of *L. latidens*. Within its current known range the beetle can be found in the full range of wet forest communities, including wet eucalypt forest, damp eucalypt forest and rainforest. Its occurrence in these forest types is unlikely, however, to be related to the floristic composition of these forests as no relationship between the beetles presence and particular wet forest communities was apparent. This is consistent with the findings of a range of studies that have investigated the relationship between floristics and invertebrate community composition (eg. Gardner 1991; Taylor *et al.* 1994; Sanderson *et al.* 1995; but see Yen 1987). Instead, it is likely to reflect a requirement of the species for a moist environment and perhaps a certain level of soil quality. However, no relationship between soil physical characteristics and the occurrence of *L. latidens* was found. This is surprising given the species' soil-dwelling habit and contrasts with the results of other studies of ground-beetles which have successfully related beetle community composition to soil properties (eg. Dufrene 1990; Luff *et al.* 1992: Niemela *et al.* 1992; Sanderson *et al.* 1995).

The occurrence of *L. latidens* in isolated patches and thin linear strips of wet riparian forest in otherwise inhospitable habitat (ie, dry forest) suggests that this species may be able to maintain viable populations in relatively small areas of habitat. Relative to upland areas, riparian areas may provide a more mesic environment (Naiman and Decamps 1997) and structurally complex habitat for many species, depending on the geomorphic setting and climatic zone in which they occur (Hancock *et al.* 1996). The intrinsic value of riparian areas as habitat for terrestrial species in otherwise unsuitable habitat is well recognised, but it has generally been studied in the context of riparian buffers retained in areas cleared for agriculture (eg. Hill 1995; French and Elliott 1999) or forest subject to clearfelling (eg. Michaels and McQuillan 1995; Taylor *et al.* 2000; Whitaker *et al.* 2000).

Preferred habitat for *L. latidens* appeared to have the following characteristics: > 10% ground cover of CWD; forest where CWD was dominated by small (< 10 cm) and medium (10-50 cm) diameter logs; and forest with a well-developed canopy cover. The preference of *L. latidens* for wet forest with these characteristics may relate to a requirement of the beetle for a relatively cool and moist micro-climate. There was some indication that *L. latidens* preferred a relatively mature forest structure with a well-developed canopy and an open understorey and the highest density sites contained forest of this type. These characteristics appear to support Michaels (1999) categorisation of *L. latidens* as a late-successional forest species rather than an old-growth specialist. Similarly, in Chapter 1 it was found that the characteristics of optimal habitat for another threatened Tasmanian lucanid, *Hoplogonus simsoni* reflected a particular forest structure rather than floristic composition of the forest. Studies of other forest-dwelling ground beetles in evergreen forest have also found that a particular forest structure influences the occurrence of different species (eg. Gardner 1991; Niemela and Spence 1994; Ings and Hartley 1999; York 1999, Magura 2002; but see Gerell 1988).

Although the quantity of fallen dead wood (CWD) on the forest floor appears to be an important component of the habitat of *L. latidens*, the species is not log-dwelling. It is a soil-dwelling beetle that prefers to inhabit the upper layer of soil underneath logs. The exact nature of this relationship between beetle, soil and logs is unknown. Research into CWD as a habitat for a huge suite of organisms has largely focussed on organisms that spend all or part of their lives living inside rotting wood (eg. Graham 1925; Savely 1939; Wallace 1953; Elton 1966; Fager 1968; Hamilton 1978; Speight 1989; Yee *et al.* 2001; Grove 2002) or those that live on the surface of logs (eg. McCullough 1948; Andersson and Hytteborn 1991; Laaka 1992; Bader *et al* 1995; McKenny and Kirkpatrick 1999; Jonsell and Nordlander 2002). However, scant attention has been directed at either the organisms that live under logs or the characteristics of this environment at the soil/log interface. Soil moisture levels under logs are likely to be higher than in surrounding soil due to the insulating effects and water-holding capacity of logs (Amaranthus *et al.* 1989), providing a habitat where the chances of desiccation are lower. This may allow *L. latidens* to inhabit the drier parts of its range. It has been demonstrated that the environment under logs may function as a refuge from disturbances such as fire (Campbell and Tanton 1981). Hence, it has

been postulated that this environment may be a significant source of propagules for the recolonisation of disturbed habitat following wildfire or clearfelling (eg. Madden *et al.* 1976; Moldenke and Lattin 1990; Michaels and McQuillan 1995). CWD may also provide an important shelter from predators such as birds. Levels of organic matter, and hence food sources for the beetle, may be higher under logs than in surrounding habitats due to fragmentation of CWD (A. Evans unpubl. data) and the leaching of exudates from the logs as they reach the latter stages of decay (Krankina *et al.* 1999). This increase in organic matter will also improve the structure of fine-textured soils and increase water-holding capacity (Lang and Forman 1978), which may be important for organisms, like *L. latidens*, living at the soil/log interface.

Conservation considerations

Suitable wet forest habitat for *L. latidens* is patchily distributed throughout its range and only 15% (42 km^2) of its range constitutes suitable habitat (Table 2). The majority of this wet forest habitat occurs in the eastern half of its mainland range, centred on Wielangta State Forest (Fig. 1). Habitat in this part of its range consists of large areas of wet forest that are generally interconnected. In the western half of the beetle's mainland range, suitable habitat is very patchily distributed and generally occurs in isolated patches and riparian corridors. It is probable that populations of *L. latidens* in these isolated patches are the most vulnerable to habitat disturbance or environmental change, as there is no means for this flightless beetle to re-colonise these areas when local populations go extinct.

Table 2. Distribution of potential habitat for *L. latidens* according to land-use category.

Land-use category	Potential habitat (ha)	% Potential habitat	% Total range
Formal reserves	780*	19%	3%
Informal reserves	470	11%	2%
State forest (couped)	2040	48%	7%
State forest (uncouped)	320	8%	1%
Private property	600	14%	2%
Total	4210	100%	15%

Source: Forestry Tasmania 31/10/2002. * Including 520 ha of potential habitat in Maria Island National Park.

With the exception of the wet forest on Maria Island National Park, suitable habitat for *L. latidens* is poorly reserved (Table 2). Twenty-seven percent of potential habitat for the species is protected in the CAR reserve system (Table 2), however, with the

exception of the extensive area of wet forest on Maria Island (12%), a substantial proportion of this protected habitat occurs as small patches or thin linear strips such as streamside reserves (~ 5%) and wildlife habitat strips (WHS) (~ 7%). Just under half (48%) of the potentially suitable habitat for the species is categorised as harvestable forest (Table 2). The majority of this corresponds to the most extensive area of wet forest habitat in the eastern half of the beetle's mainland range.

It was beyond the scope of this study to examine the effects of disturbance on *L. latidens* in any detail, however there was sufficient evidence to suggest that the selective logging practices of the past have not severely impacted on the species, with over half of the recorded localities showing signs of partial logging. Unlike the situation for other threatened lucanids in Tasmania (Chapter 1; Chapter 4), little potential habitat of *L. latidens* on State forest is threatened by conversion to plantation, with less than 1% potentially subject to this practice in the future (Forestry Tasmania, unpubl. data). The majority of wet and damp eucalypt forest coupes (averaging 50 ha in size) within the range of *L. latidens* are currently harvested by clearfelling often followed by a high intensity regeneration burn and aerial sowing - a clearfell, burn and sow or CBS silvicultural regime (Forestry Commission 1994). Current sustainable yield models indicate that the regenerating forest may be subject to harvesting again at approximately 90-year intervals (Forestry Tasmania 2002). The effects of such practices on populations of the beetle are largely unknown. However, Michaels and Bornemissza (1999) suggested that *L. latidens*, and other CWD-associated lucanids in Tasmania, will suffer long-term population declines in landscapes where CBS forestry activities predominate due to the depletion of CWD over successive harvesting rotations. In addition, Forestry Tasmania has adopted a policy of "more timber from less land" (Forestry Tasmania 1998a) through the use of more intensive forest management practices such as regrowth-thinning, to maximise the timber-producing capacity of regenerating forest throughout a silvicultural cycle (Forestry Tasmania 2001). Although these practices are not currently used extensively within the range of *L. latidens* (Forestry Tasmania, unpubl. data), it is likely that in the future (ie, after the first harvesting rotation) they will be. A significant proportion of potential habitat of *L. latidens* occurs on private land (Table 2), and hence has an uncertain future due to the vagaries of land-owner intent. Over the last decade in particular, a proportion of potential habitat on private land has already been converted to short-rotation pulpwood

plantations (Forest Practices Board unpubl. data), a practice that other studies have suggested may lead to local extinctions of species such as *L. latidens* (Chapter 1; Chapter 4). Conservation measures to supplement the current reserved areas of potential habitat are clearly required to ensure the persistence of *L. latidens* into the future.

In the assessment of threatening processes and the development of any conservation measures for a species, the degree to which the effects of various land-use practices on habitat deviate from those produced by natural disturbances needs to be considered (Lindenmayer and Franklin 2002). As wet forest types constitute potential habitat of *L. latidens*, it is clear that the species has had evolutionary experience of severe disturbances such as wildfire. Wet eucalypt forests are naturally fire-prone, and wildfires of various scales and intensities, with a return interval of 20-400 years (Hickey *et al.* 1999; McCarthy *et al.* 1999; Alcorn *et al.* 2001) are thought to be the main source of large-scale natural disturbance. However, many elements of CBS silviculture, the main forestry practice within suitable habitat of *L. latidens*, are qualitatively different from the effects of wildfire, including the resultant forest age structure and the landscape-scale spatial pattern of disturbance (McCarthy and Burgman 1995; McCarthy and Lindenmayer 1998; McCarthy *et al.* 1999). Given the stochastic nature of wildfires, CWD levels will naturally fluctuate in space and time. Modelling of CWD dynamics under various forest management scenarios in Tasmania's wet eucalypt forest by Grove *et al.* (2002) would appear to support the concerns of Michaels and Bornemissza (1999) regarding the impacts of CBS and other intensive forestry practices on CWD-dependent biota. Grove *et al.* (2002), in a comparison of CBS with a wildfire scenario, found that the main issues for CWD-dependent biodiversity in areas subject to CBS were: significant depletion in CWD levels over successive 80-90 year rotations, particularly of large diameter logs; long-term reduction in small-scale connectivity of CWD; and the establishment of single-aged forest CWD dynamics where multi-age dynamics may be closer to the natural situation. They concluded that intensive forest management practices, such as conversion of native forest to plantation, regrowth thinning, and fuelwood harvesting following clearfelling, were likely to exacerbate these impacts (Grove *et al.* 2002).

Whilst activities such as fuelwood harvesting are not planned within the range of *L latidens*, many coupes within the range of the species are available for firewood harvesting following clearfelling, resulting in a further depletion of CWD resource at the local scale. Although permits for harvesting particular quantities of firewood are required, there is minimal monitoring of amounts extracted and no strategic planning in terms of resource availability or potential environmental impacts. Driscoll *et al.* (2000) estimated annual household consumption of firewood in Tasmania at 720,000 tonnes, including 50,000 tonnes for which permits were issued for State forest. This latter figure is likely to be a significant underestimate of the amount of firewood sourced from State forest, given the widespread nature of illegal firewood collecting (Driscoll *et al.* 2000). Firewood collecting has been identified as a threatening process for a suite of species including nine mammal species, 22 bird species, and 24 reptile species Australia-wide (Driscoll *et al.* 2000). Although limited in scale, the potential impact of firewood collecting on a CWD-dependent species such as *L. latidens* lies in the cumulative nature of the impact, when it follows forestry activities such as CBS.

As no "hotspots", areas containing high-density populations of *L. latidens*, or optimal habitat were identified in this study, it is not possible to recommend further reservation of particular areas within the beetle's range. Hence, the conservation of this species will largely depend on existing reserves, both formal and informal, and a multi-scaled approach to the maintenance of contiguous areas of suitable habitat in 'off-reserve' areas throughout its range. At the landscape scale, the function of these areas is to ensure that populations in relatively undisturbed areas do not become isolated and to increase the chances of genetic exchange between populations. These areas will also act as sources of individuals to recolonise regenerating forest if, or when, they become suitable habitat for the beetle. The system of Wildlife Habitat Strips (WHS) and streamside reserves established throughout production forests in Tasmania (Taylor 1991) provide a foundation on which such a network can be built. The current bias in the location of these strips towards streams will advantage *L. latidens*. However, given the relatively infrequent occurrence of WHSs there is a need to maximise the retention of potential wet forest habitat at a smaller (or forest stand) scale. This could be achieved through widening current streamside reserves and extending their application to smaller streams and drainage lines containing potential habitat. Maximising the retention of potential beetle habitat in this way will

be critical in the drier parts of the beetle's range where areas of suitable habitat are largely restricted to riparian corridors. At the local (or coupe) scale, the retention of CWD should be maximised, and wherever possible, silvicultural practices that involve some within-coupe retention of standing timber should be employed to ensure recruitment of CWD into the future.

For a species dependent on a particular forest structure and a dynamic habitat such as CWD, a relatively static habitat-retention approach, as outlined above, is likely to be insufficient to ensure the long-term conservation of *L. latidens* throughout its range. Research into the natural dynamics of forest structural elements such as CWD in Australia is very much in its infancy. There is an urgent need for further research into the natural fluctuations of CWD in space and time and the responses of CWD-dependent biota, including threatened species such as velvet worms (Mesibov and Ruhberg 1991) and lucanids (Chapter 4), under various disturbance scenarios as outlined by Grove *et al.* (2002).

Conclusions

This study found that *L. latidens* is a soil-dwelling lucanid that occurs in wet and riparian forest at very low population densities and whose occurrence is related to a mature forest structure, and in particular the presence of CWD. Despite the range extensions recorded here, the species meets the criteria of an Endangered taxon under the Tasmanian *Threatened Species Protection Act* 1995 due to its restricted distribution, its occurrence at very low population densities, severe fragmentation of populations, the inadequate reservation of potential habitat, and the likely negative impact of current forestry practices. The results of this study suggest that the conservation of this species in 'off-reserve' areas on mainland Tasmania is best served by maximising the retention and linkages of unlogged wet forest habitat throughout its range, coupled with adaptation of silvicultural practices to ensure 'in-coupe' retention of suitable habitat into the long-term. The conservation requirements of *L. latidens* illustrate the inadequacies for forest-dependent threatened species of conservation management strategies that focus on reserve creation. They also illustrate the need to expand the focus of spatial-based or static conservation management strategies to incorporate the temporal dynamics of habitat maintenance.

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CHAPTER 4

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DISTRIBUTION AND CONSERVATION STATUS OF THE MT MANGANA STAG BEETLE *LISSOTES MENALCAS* (COLEOPTERA: LUCANIDAE)

Jeff M. Meggs and Robert J. Taylor

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ABSTRACT

Lissotes menalcas (the Mt. Mangana Stag Beetle) is an obligate log-dwelling beetle and is presently classified as Vulnerable under the Tasmanian *Threatened Species Protection Act* 1995. The distribution, habitat and conservation status of this species were investigated through a combination of collation of known locations and searches for the species in decaying logs across likely areas of occurrence. *Lissotes menalcas* was found to have a wide distribution in south-eastern Tasmania and is now recorded from 34 localities. The species was found in a variety of wet forest habitats from old-growth mixed forest through to 23 year old wet eucalypt silvicultural regrowth. About 12% of wet eucalypt forest within the predicted range of *L. menalcas* is formally reserved and another 14% on Crown land is unlikely to be subject to logging. *L. menalcas* appears to tolerate both wildfire and clearfelling and regeneration burning. Because the species depends on an ongoing supply of rotting logs for its long-term survival, plantation development will most likely lead to the elimination of the species from such areas. Analysis of the likely impact of the plantation establishment within its range over the next 10 years indicates that it will not reach levels that would lead it to be considered as Vulnerable. It is thus recommended that the status of the species be changed from Vulnerable to Rare. However, there is a need to determine the ecological sustainability of present forest management practices in relation to the decaying-log habitat and to continue to monitor the extent of clearing and modification of the species' habitat.

Keywords: Lissotes menalcas; stag beetle; threatened species; conservation; plantations; wet eucalypt forest; Tasmania.

INTRODUCTION

The Mt. Mangana Stag Beetle *Lissotes menalcas* Westwood is a large, black, flightless beetle, the males of which have large mandibles (Pl. 1). Its body length varies between 16 and 25 mm. It is endemic to Tasmania and readily distinguished from other Tasmanian lucanids by the following features: a highly polished exoskeleton, causing the males to appear as if coated with black enamel; a peculiarly

shaped prothorax with a humped or peaked appearance; and a largely excavated head (Lea 1910). The female is also highly polished but not as shiny as the male due to the much denser and larger punctures on the head and prothorax. *L. menalcas* is a relatively long-lived beetle (five to six years, including larval stages) (G. Bornemissza pers. comm.). It is believed to spend the majority of its life within decaying logs feeding on wood-rotting fungi.

Plate 1. *Lissotes menalcas* adult male, dorsal view (Body length is 2 cm excluding the mandibles).

Lissotes menalcas was first collected prior to 1855 (Westwood 1855). Previous to this study it was only known from a handful of sites in south-eastern Tasmania (Jackson and Taylor 1995). *L. menalcas* was classified as Vulnerable under the Tasmanian *Threatened Species Protection Act* 1995 due to the small number of recorded localities and its apparent disappearance from some of these localities, not having been recollected from these since early this century.

Meggs (1996) has shown that first rotation logging has little influence on the size or quantities of decaying logs or the types of rot present. However, with successive rotations on an 80 year or shorter cycle it is likely that the volume and size of logs available would decline. Studies in Europe and North America of sites which have been subject to intensive forest operations over several rotations have documented a dramatic reduction in the volume of rotting-log resource, particularly large diameter logs (Gore and Patterson 1986; Andersson and Hytteborn 1991) and late-stage rotting

logs (Bader *et al.* 1995) which are important habitat for a number of beetle families including Scarabaeidae, Lucanidae and Passalidae (Harmon *et al.* 1986).

Given the cryptic habits of the species it was suspected that it was probably more widespread than indicated by the available records. However because of its log-dwelling habit and the potentially high proportion of its range occurring in wood production forests the species could be at risk. The present study was thus undertaken to reassess its distribution and conservation status.

METHODS

Records of the species were collated from museum and private collections. Field work was conducted in south-east Tasmania between November 1995 and April 1996. and in May 1998 at 43 sites. All sites searched were in wet eucalypt or mixed forest. These forest types were targeted so as to maximise our chance of finding the species in an area. We did not test our assumption that the species does not occur in dry forests but felt this was unlikely given that all museum and private records were from wet forest.

Because of its cryptic habit, the sampling method used to locate *L. menalcas* was destructive and involved breaking open logs and searching the interior for adults. Logs lying on the ground as well as elevated logs of all sizes were searched. The only selection criterion used was that the exterior of the log exhibited signs of decay, including visible rot at the end or at a split, or a well-developed moss covering suggesting that the log had been down for some time. More intensive searching was undertaken in those logs which contained moist red-rot, a type of wood-decay which *L. menalcas* was thought to prefer (G. Bornemissza pers. comm.). If white rot was encountered, the entire log was not searched, only the section that was broken. White rot tended to be either very fibrous or very wet and pulpy and hence tended to fall apart and did not require the detailed searching that red rot did. Sites were searched until *L. menalcas* was found or two hours had elapsed. Because of the destructive sampling method no searches were permitted in reserves controlled by the Parks and Wildlife Service.

Traps were set at two known localities of *L. menalcas* in the southern forests to determine whether standard pitfall trapping was a reliable sampling method for this species. Each trap consisted of a plastic cup (diameter of 85 mm) with ethylene glycol as a preservative placed inside a buried section of PVC pipe so that the rim of the cup was flush with the soil surface. Traps (16 at one site and 20 at the other) were set at least 2-3 m apart as close as possible to decaying logs. Traps were left in place for 12 weeks from early December and were emptied at six weekly intervals.

Habitat data (vegetation, leaf litter, soil, geology, numbers and sizes of logs, slope, aspect, altitude, presence of fire scars on standing or fallen trees and any evidence of logging) were collected at all known localities of *L. menalcas* with the exception of those that have been logged or burnt since the specimen was recorded (two sites), those without precise locality details (four sites), or to which access could no longer be obtained (one site).

The environmental modelling package CORTEX (Brereton *et al.*. 1997) which has been derived from BIOCLIM (Busby 1991) and GARP (Stockwell 1993) was used to predict the potential area where suitable environmental conditions occurred for *Lissotes menalcas*. Ten environmental variables determined at a scale of 1 km grid squares were used in the model: two temperature, two rainfall, soil nutrient level, slope, aspect, topographic position, and elevation. The model constructs environmental envelopes from combinations of ranges and/or categories of variables which imply the presence of the species. This analysis was undertaken in the Parks and Wildlife Service GIS section. The predicted range of the species was determined based on the following: the results of the cortex model, the proximity to known locations, the presence of known sampling areas and the extent of suitable habitat present. The results of the CORTEX model were only used as one of the guides to predicting the distribution as factors other than those used in the model may determine a species distribution. It is not uncommon for an invertebrate species not to occur in parts of the environmental range predicted by a CORTEX model when these are searched for the species (Meggs 1999)

RESULTS

Distribution

Lissotes menalcas was found to have a wide distribution in south-eastern Tasmania (Fig. 1). In addition to the six localities known prior to this study (Jackson and Taylor 1995), a further 12 records were obtained from private collectors and institutions, and another 16 were obtained by searching. *L. menalcas* has now been recorded at 34 localities with an additional unconfirmed record from Ida Bay. This record was obtained from a list of Tasmanian Lucanidae produced by Barry Moore and provided to George Bornemissza. Locations were given but grid references were not included. The discovery of this species on the Tasman and Forestier Peninsulas has extended its range east by just over 40 km representing a significant increase in its known distribution (Fig. 1).

The extent of occurrence of *Lissotes menalcas*, measured as the land area within the smallest convex polygon which contains all the sites of occurrence (calculated separately for the southern forests, South Bruny and the Tasman and Forestier Peninsulas combined) is 1831 km^2. The CORTEX model (Fig. 2) showed suitable environment for the beetle occurring in many areas well to the north and south of the known locations. Our more conservative predicted extent of occurrence (Fig. 2) was 3068 km^2.

Pitfall trapping proved to be an unreliable method for sampling *Lissotes menalcas*. During a 12 week period with a total of 36 traps laid at two localities at which the species was known to occur only one female *L. menalcas* was trapped.

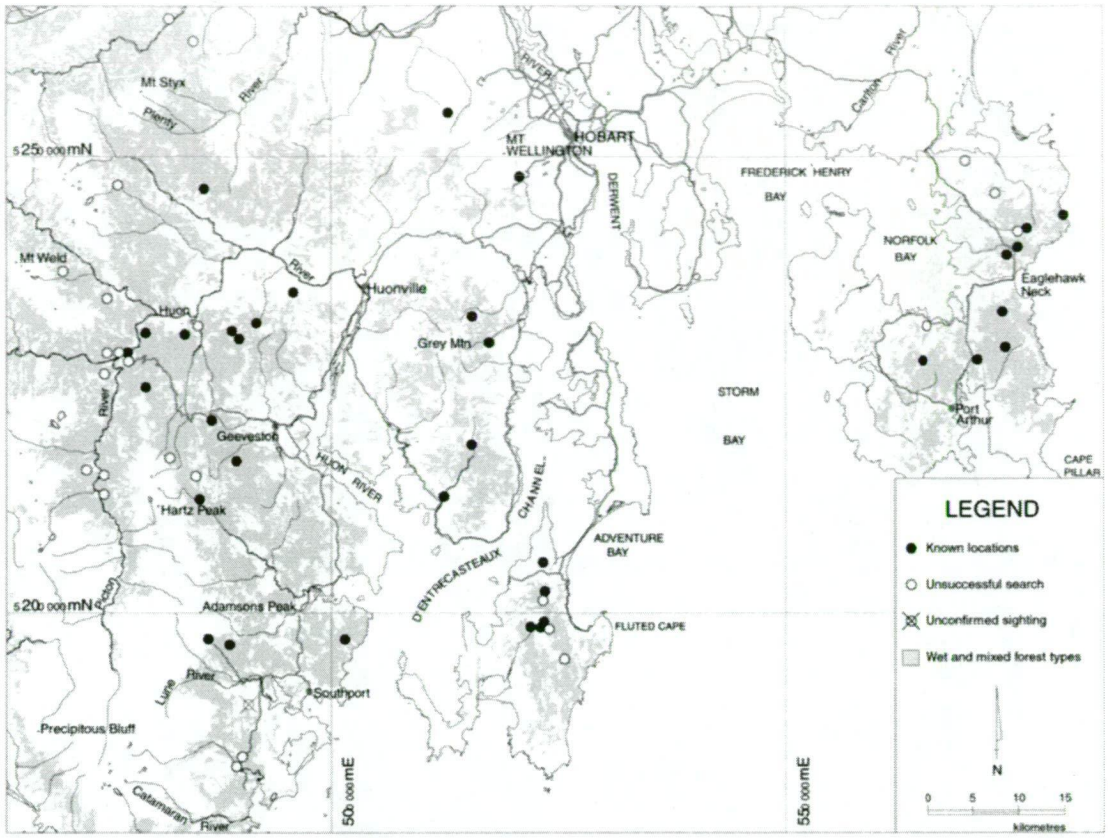

Figure 1. Known locations for *Lissotes menalcas*, sites of unsuccessful searches during this study and distribution of wet eucalypt forest in south-east Tasmania.

No data are available on population sizes. Three areas of high population densities in the Southern Forests were reported by private collectors. At these locations up to 10 to 12 *Lissotes menalcas* have been found in a single log (G. Bornemissza, M. Bouffard, and A. Vanderschoor, pers. comm.).

Habitat requirements

At sites where the species occurs, annual rainfall varies from just under 700 mm (on the Forestier Peninsula) to just over 1220 mm (in the Wellington Ranges). *Lissotes menalcas* was found at altitudes up to 600 m but no sites over 650 m were searched in this study, although suitable vegetation occurs up to approximately 850m. Slightly less than half (15 out of 34) of the recorded localities of *Lissotes menalcas* occur on Jurassic dolerite. The remaining records occur on a variety of sedimentary rock types. Soils were generally clay loams.

Figure 2. Predicted distribution of suitable environmental conditions for *Lissotes menalcas* in Tasmania from the CORTEX environmental modelling package. The darker the shade of grey the higher the predicted suitability of the environment. The black squares are known locations. Further sites of predicted suitability occurred in more northern areas of eastern Tasmania which are not shown. The black line represents the proposed limits of the predicted distribution in the mainland region.

Leaf litter composition and depth varied greatly both within and between sites. *Lissotes menalcas* was found in both wet eucalypt forest and mixed forest, but was found more frequently in the former (70% compared with 35%; χ^2=6.3, p<0.05). Sites differed markedly in floristic and structural composition and successional stage (ranging from even-aged wet eucalypt regrowth forest to mixed-age, late successional mixed forest). Sites were dominated by either *Eucalyptus regnans, E. globulus* or *E. obliqua*. *L. menalcas* was found in all three of the natural regrowth sites searched and three of the four silvicultural regrowth forest sites.

During this study one female was found in white, pulpy rot inside a sassafras *Atherosperma moschatum* log. All other *L. menalcas* found occurred in eucalypt logs, either just below the surface in slightly rotten logs or deep within the core of well-

rotted logs. Generally the rot was a dark red and had a moist, clay-like consistency. This rot type does not appear to occur in rainforest trees. There was some variation in colour and wetness of the rot, from less moist orange-red rot through to saturated brown humus-like rot. However, only dead specimens were found in the latter. *L. menalcas* has also been collected from myrtle *Nothofagus cunninghami* logs (G. Bornemissza, pers. comm.) and from under wood debris in a sawdust mound at the site of an old timber mill (A. Vanderschoor, pers. comm.).

Three of the recorded localities were severely burnt in the late 1960s by wildfire which completely removed the canopy species present. At one of the sites, even small drainage lines and gullies appear to have been burnt, discounting the possibility of *L. menalcas* surviving in unburnt refuges. The silvicultural regrowth forest sites where *L. menalcas* was present would have been regenerated by a high intensity burn.

Distribution by land tenure and land use

Twenty three (68%) of the 34 known locations for *L. menalcas* occur on land presently under the management of Forestry Tasmania. Seven of these are in areas which are, or are soon to become, dedicated reserves (Abel Tasman Forest Reserve, Tahune Forest Reserve, Mt. Mangana, Hastings Caves - Southwest National Park extension, Lonnavale 1). Of the other known locations managed by Forestry Tasmania nine are planned for wood production, one of which has been logged since the species was found there, and the remaining seven are in management zones protected from logging operations. Eight sites occur on private property. The remaining three sites are in reserves managed by the Department of Primary Industries, Water and Environment (Wellington Park, Snug Falls Nature Recreation Area).

The percentage of the habitat which is under different tenures or projected land uses is roughly comparable in the three major regions inhabited by the species (mainland, South Bruny, Tasman and Forestier Peninsulas) and so only the overall figures are presented in Table 1.

Table 1. The percentages of wet eucalypt forest and rainforest present in different land tenures and land use categories for the range of *Lissotes menalcas* based on the extent of occurrence and the predicted distribution and the area of these vegetation types in the three major regions in which the species occurs.

	Wet eucalypt		Rainforest	
	Known extent	Predicted extent	Known extent	Predicted extent
PERCENTAGE OF HABITAT				
Crown land				
formal reserve	11	12	46	65
not to be logged	13	14	29	20
production forest	51	49	23	14
Private land (%)	25	25	2	1
AREA OF HABITAT (km^2)				
mainland	671	867	21	43
South Bruny	2	72	0	<1
Peninsulas	45	193	1	1
Total	718	1132	22	45

Note: The extent of occurrence is the smallest convex polygon encompassing all of the known locations. This was undertaken separately for the three regions. The predicted distribution is based on the known locations, the extent of suitable habitat and the predicted occurrence of suitable environments from the CORTEX environmental modelling software. The area of Crown land not to be logged is made up of areas of State forest which are either informal reserves or have not been placed into coupes which are earmarked for production forestry.

The figures for rainforest are presented separately from wet eucalypt forest as, although some of these areas will support *L. menalcas,* it is unlikely that densities would be as great due to the apparent preference of the species for eucalypt logs. Hence, the values for the wet eucalypt forests (which include sites with a rainforest understorey) are most relevant to assessing the amount and likely future of habitat of the species. These figures are derived from the vegetation mapping undertaken for the development of the Tasmanian Regional Forest Agreement and are applicable as at June 1996. Prior to this date 35 km^2 from the area of occurrence and 48 km^2 from the predicted range had been converted to plantation on State forest and hence would not have been counted in the area of habitat figures in Table 1. All of these plantations have been established on sites formerly supporting wet forest. Between the time of the production of the forest vegetation map for the *Regional Forest Agreement* (June 1996) and March 1999, for wet forest habitat, 431 ha within the area of occurrence and 905 ha within the predicted range was converted to plantation. Thus, to date plantation development has reduced the area of wet eucalypt forest on Crown land by 5.4% in the area of occurrence and 5.0% of the predicted range from

the Crown land estate. No estimates are available on how much habitat has been cleared from private land. *Lissotes menalcas* has 134 km^2 (12%) of wet eucalypt forest habitat reserved across its predicted range and a further 161 km^2 (14%) in State forest which is unlikely to be subject to logging.

DISCUSSION

Distribution

This study has substantially extended the known range of *Lissotes menalcas* in south-east Tasmania to the Forestier and Tasman Peninsulas. The rediscovery of *L. menalcas* in the Wellington Ranges is important as it had not been recorded there since early this century (Lea 1910). Together, these findings have increased the known range of the beetle by more than 25%.

The distribution of *L. menalcas* may be greater than is currently known. The environmental domain for the species predicted from the CORTEX model indicated that a suitable environment exists in many areas outside the current known range. However, this study presents a conservative sub-set of this domain as a potential distribution for the species. This predicted distribution needs to be tested by further sampling of areas that the CORTEX model deemed to be environmentally suitable which we have not included.

On South Bruny Island and the Tasman and Forestier Peninsulas all of the predicted areas were considered to be likely for *L. menalcas*. The predicted southern mainland boundary was conservatively taken to be the D'Entrecasteaux River based on an unconfirmed record from Ida Bay. However, a suitable predicted environment and habitat is present south of the D'Entrecasteaux River and further searches could possibly find the species in this area. The predicted western boundary was determined on the basis of the known locations, the presence of suitable habitat and the climatic predictions. The predicted northern boundary was based on the known locations as there was no evidence provided by the climatic model that they would occur further north in that particular region.

It is unlikely that the species occurs in the Wielangta area, as predicted by the CORTEX model, as much searching conducted by several investigators (J. Meggs, K. Michaels, G. Bornemissza, M. Bouffard, G. Blake unpubl. data) has failed to reveal its presence. However, the northwestern limit delineated in Fig. 2 could be extended with further searching. Although two searches in the Styx and West Uxbridge areas failed to locate the species, the Florentine and Mt. Field areas remain to be searched.

To our knowledge only one specimen additional to the one captured in this study has ever been caught in a pitfall trap. This was by R. Bashford, (pers. comm.) in 1996 near the Kermandie River in the same type of pitfall used in our study. The very low capture success of the pitfall traps indicates that pitfall trapping is not a suitable means of sampling populations. In a study of the threatened stag beetle *Lissotes latidens* in eastern Tasmania Michaels (1996) also found lucanid species that were not captured in pitfall traps by searching for them in logs. Direct sampling appears to be the most reliable method available to locate *L. menalcas*. Unfortunately, the log-dwelling habit of *Lissotes menalcas* makes direct sampling a destructive, time consuming and serendipitous sampling method.

Habitat requirements

Lissotes menalcas was found in a broad range of wet forest communities ranging from old-growth mixed forest to 23 year old wet silvicultural regrowth forest. Because it was also found in rainforest species logs, albeit at lower rates than in eucalypts, it is highly likely that it also will occur in rainforest. Because *L. menalcas* is an obligate log-dweller, its occurrence and population density will be influenced by the size and abundance of logs and the extent and type of log decay. Further research is required to elucidate the particular microhabitat within decaying logs preferred by *L. menalcas*.

In wet forests that *L. menalcas* inhabits, clearfelling followed by an intense regeneration burn is the standard silvicultural treatment. The species appears to be able to survive this type of disturbance. In the sites with silvicultural regrowth three of the four logs in which *L. menalcas* were located would have been present on the forest floor prior to logging. Most of the logs present as a result of logging did not

appear to have decayed enough to have produced suitable habitat for *L. menalcas*. The protection afforded by the log habitat, at least in the larger logs, may be sufficient to insulate the species not only from the destructive effects of fire but also the drying effects of the absence of a forest canopy in the years it takes a substantial canopy of regrowth to develop. However, the possibility of reinvasion of the logged area following disturbance from logging and burning cannot be ruled out. Anecdotal evidence that log-dwelling fauna survive burning is provided by Mesibov (1988, p.20): "In the course of my fieldwork I broke open literally hundreds of old eucalypt logs with jagged, burned ends and a hard, blackened surface, to find inside a mass of moist, unburned wood rot and a teeming community of invertebrates." Fauna in the deep layers of soil are less effected by regeneration burning after timber harvesting than those in the litter (Vlug and Borden 1973). The importance of refuges such as under logs and stones to species survival in burnt areas has also been demonstrated (Madden *et al.* 1976). The interior of decaying logs may offer another such refuge. This may explain why the log dwelling fauna of older regrowth and old-growth are similar (Taylor 1990).

Plantation development in native forests will most likely lead to the local extinction of *L. menalcas* populations due to the virtual elimination of rotting logs. Populations may survive in the short term in windrows (Bashford 1990). However, by the second rotation most of this resource has rotted away, with little to no replacement of the log resource from standing timber. Specific prescriptions for the sustainable management of the rotting-log resource are likely to be required where large areas of native forest are converted to plantation in order to limit the impact of such practices on *L. menalcas* and other organisms associated with decaying logs.

Conservation status

Lissotes menalcas is presently classified as Vulnerable under the Tasmanian *Threatened Species Protection Act* 1995. The Tasmanian Threatened Species Advisory Committee has released criteria for use in assessing the conservation status of species (Threatened Species Advisory Committee 1998). These are similar to those adopted by the IUCN except that Tasmania uses an additional category, Rare. If these criteria are applied to the current situation, the species does not warrant listing

as Endangered or Vulnerable but does qualify as Rare because of the size of its range (less than 100x100 km) combined with the fact that threatening processes (plantation development and clearing of forest on private land) are operating. Some of the criteria for assessing conservation status relate to the predicted extent of loss of habitat over the next 10 years. If a species will undergo a population decline of at least 20% within the next ten years it is considered to be Vulnerable. Forestry Tasmania has adopted a policy of increasing its rate of plantation establishment on State forest to compensate for the loss of resource following tenure changes under the Tasmanian Regional Forest Agreement. Current planning estimates from Forestry Tasmania indicate that it is highly unlikely that more than 10% of wet eucalypt forest within the area of occurrence or within the predicted range will be converted to plantations over the next ten years. There is no indication of the future extent of loss of habitat on private land. However the figure in Kirkpatrick (1991) indicates that clearing within the range of *L. menalcas* in the period 1980-88 was way below levels necessary to achieve a 20% reduction in *L. menalcas* habitat within the next 10 years. It is thus recommended that the status of the species under the Tasmanian *Threatened Species Protection Act 1995* be changed from Vulnerable to Rare. However, given the restricted range of *L. menalcas*, the impact of clearing and modification of habitat on the species should be continued to be monitored. This is particularly the case for private land where, unlike the situation on Crown land, no formal overview is being conducted. There is also a need to determine the longer term ecological sustainability of native forest logging on the decaying log habitat. It is presently unknown whether such habitat will be maintained after successive rotations of logging of native forest. Further assessment is also required of the effects of thinning of regrowth, a method increasingly being used to boost sawlog production from native forests.

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GENERAL DISCUSSION

GENERAL DISCUSSION

A fundamental requirement for the development of conservation management strategies for a threatened species is knowledge of the spatial distribution and extent of habitats utilised by the species and the identification of areas where the conservation requirements of the species may conflict most strongly with land-use practices. This was achieved to varying degrees for each of the three stag beetles, *Hoplogonus simsoni*, *Lissotes latidens*, and *L. menalcas* studied here. All three species were more widely distributed than previously thought, and in terms of broad forest types can be considered wet forest species. Despite the increases in their known ranges, there remains a relatively low coincidence between the distribution of potential habitats for each species and the CAR reserve system, and a relatively high coincidence with areas currently or likely to be subjected to intensive forestry practices. Hence the conservation requirements of the three species will largely need to be met by off-reserve mechanisms for habitat conservation, complemented by existing reserved habitat.

The degree of success in determining the distribution and habitat requirements of each species was partly dependent on the problems involved in establishing their specific habitat characteristics, including difficulties associated with sampling cryptic species, the availability of relevant GIS data, and, to some extent, the availability of funding for research into each species. The detailed study of *H. simsoni*, presented in Chapters 1 and 2, represented a successful transfer of conservation biology techniques largely developed for vertebrate conservation, such as predictive habitat modelling (eg. Munks 1993; Lindenmayer *et al.* 1995; Mladenoff *et al.* 1995; Mace *et al.* 1999; Pearce & Ferrier 2001), to a threatened invertebrate. It was possible to identify with a reasonable level of confidence the spatial distribution of quality habitat for *H. simsoni* down to a scale of 1 ha patches of forest. This was partly facilitated by an efficient means of sampling relative population sizes, enabling the identification of hotspots of abundance, the availability of vegetation and topographic GIS data at an appropriate scale, and a level of research funding roughly four times higher than was available for the other two species. Conversely for the two *Lissotes* spp. no hotspots of abundance could be identified and no GIS data existed for the particular structural elements identified as important components of their habitat (Chapters 3 and 4). Thus, only broad vegetation

data could be used to predict the spatial extent and conservation status of potential habitat. It is possible that these species occur at relatively low population densities across their ranges, however, without a more efficient method for sampling CWD-dependent fauna this remains unknown. In the case of *L. menalcas*, it was not possible to quantitatively assess relative population sizes. This was because a condition of the research permit issued under the Tasmanian *Threatened Species Protection Act* (TSPA) 1995 was that sampling must cease after location of the species due to the destructive nature of the sampling method used (Chapter 4). Nevertheless, for these species, the coarser-scale predictive modelling package, CORTEX, proved a valuable tool in predicting potential habitat for these species beyond the sites sampled. Importantly, predictions for all three species were evaluated, as recommended in other studies (Fielding & Bell 1997; Manel *et al.* 1999; Guisan & Zimmermann 2000). This was done either by field-checking, in the case of *H. simsoni* (Chapter 2) and *L. latidens* (Chapter 3), or, in the case of *L. menalcas*, by reference to existing knowledge of sites of species' absence (Chapter 4). Hence, land managers can have a reasonable level of confidence in the range boundaries and the distributions of habitats for each species presented.

The importance of a particular forest structure to the occurrence of *H. simsoni* and *L. latidens*, and of structural legacies such as CWD to the two *Lissotes* spp., rather than the floristic composition of vegetation is consistent with the finding of other studies (eg. Gardner 1991; Mesibov and Ruhberg 1991; Niemela and Spence 1994; Ings and Hartley 1999; York 1999, Magura 2002; but see Gerell 1988). This illustrates a need to incorporate the temporal dynamics of habitat into conservation strategies for invertebrates, and to shift from a static, set-aside approach to the conservation management of species with dynamic habitats. Brown (1996) has highlighted the dangers of managing protected habitat of threatened species, such as the orange bellied parrot and King Billy pine, under a philosophy of benign neglect in Tasmania's formal reserve system. For the three stag beetles, if natural disturbance regimes can no longer operate due to management of other forest values in both reserve and off-reserve areas, it may be necessary to actively manage important habitat for the species to ensure the continuity of these structural attributes into the future. A key component of this will be gaining an understanding of the dynamics of CWD in natural and managed forests (Grove *et al.* 2002).

Many aspects of the biology and ecology of the three stag beetles remain to be investigated. With the exception of *L. menalcas*, the species do not neatly fit into a particular functional group, though all three species have some association with CWD. Coarse woody debris appears to provide a shelter and possibly an over-wintering habitat for *H. simsoni* (J. Meggs pers. obs.), however, the larval stage of this species is soil-dwelling and larva have been found in soil away from logs (J. Meggs pers. obs.). During the active season, adults appear to be free-living, wandering amongst leaf litter. Similarly, *L. latidens* can be considered a leaf litter species or ground-beetle during the active season, but both larvae and adults are more commonly found at the soil/log interface under logs. This interface may constitute a distinct habitat, with its own 'hypoxylic' fauna.

This lack of knowledge about the biology and ecology of the three stag beetles and their responses to habitat disturbance significantly hampers their conservation. Given their occurrence in wet eucalypt forest, it is clear that they have evolutionary experience of severe disturbances such as wildfire. Whilst all three species have been recorded from young silvicultural regeneration following CBS, there were insufficient data to draw any firm conclusions on the long-term effects of such practices on populations of the species. Hence it is important to examine the degree to which the effects of forestry practices such as CBS on habitat deviate from those produced by natural disturbance (Lindenmayer and Franklin 2002). In the case of *H. simsoni*, the occurrence of the species in high numbers in 70 year old wildfire regrowth led to the conclusion that the species may recover from native forest harvesting practices such as CBS within proposed 90 year rotations, but will be eliminated if clearfelling is followed by establishment of pine plantation (Chapter 1). With regard to the CWD-dependence of the two *Lissotes* spp., it is well established that practices such as CBS will result in the depletion of CWD over successive rotations, in the absence of any mitigation measures (eg. Gore and Patterson 1986; Harmon *et al.* 1986; Spies *et al.* 1988; Andersson and Hytteborn 1991; Bader *et al.* 1995; Grove *et al.* 2002; Lindenmayer *et al.* 2002). Whilst we can be reasonably confident in making predictions on the likely effects of such practices on habitats at the local scale, scaling these effects up to the landscape is much more difficult. Yet it is at the landscape-scale that species' decline and/or extinction will manifest itself (eg. Niemela 1997; Grove *et al.* 2002). For example, *L. menalcas*

appears to have a preference for CWD containing red-rot (Chapter 4) and Yee *et al.* (2001) have found that this rot-type is generally associated with large diameter logs. Under current proposed rotations (Forestry Tasmania 2002), large logs will be eliminated in areas subject to CBS practices over successive rotations. Therefore, the long-term survival of the species in areas where such practices predominate will depend on the extent to which CWD continues to feature in the landscape (Grove *et al.* 2002) in set-asides such as wildlife habitat strips and streamside reserves (Taylor 1991). The recently conducted Population Viability Analysis for *H. simsoni* under various forest management scenarios (Fox, Meggs, Munks, McCarthy and Burgman unpubl. data), and that proposed for *L. menalcas* (Grove *et al.* 2002) should make a valuable contribution to our understanding of the landscape-scale effects of forestry on these species.

***Development of conservation strategies for the* H. simsoni, L. latidens *and* L. menalcas**

The habitat and conservation requirements of the three species evaluated in this study suggest that the multi-scaled approach to the conservation of forest-dependent biodiversity as advocated by Lindenmayer and Franklin (2002), largely based on knowledge of vertebrate responses to habitat disturbance, is just as relevant to invertebrates. Since urgency for action was one of the main drivers for the studies of the three stag beetles, much of the research presented here has already been used in the development of conservation strategies for the species by land managers and conservation agencies (Forest Practices Board 2001b). The overall aim of the strategies developed for all three species was to maintain viable populations of the species throughout their ranges by maintaining connectivity between populations through the retention of contiguous areas of suitable habitat. The varying degree of comprehensiveness of the strategies is largely a consequence of the varying success with which the characteristics of habitat utilised by each species were determined, and the lack of quantitative information on the responses of the species to habitat disturbance.

The predictive habitat model developed for *H. simsoni* (Chapter 2) has been used as the basis of a comprehensive management plan for the species on State forest (Forest Practices Board 2001b), to supplement habitat captured in the CAR reserve system. This plan includes a moratorium on logging activities within two contiguous areas in the

eastern part of the species' range, encompassing approximately 500 ha of optimal and sub-optimal habitat. Plantation establishment has been excluded from all other patches of predicted optimal habitat, and a 10% limit has been placed on the amount of potential habitat throughout the species' range that may be converted to plantation. The 10% limit was established so that, at a minimum, the conservation strategy should ensure that the species no longer meets the '20% future habitat loss' criterion for a Vulnerable taxon (Scientific Advisory Committee 1998). Native-forest logging, including CBS silviculture, can continue to be practiced throughout production forest areas within the species' range, as the limited evidence available (Chapter 1) suggested that populations may fully recover within current planned rotation lengths. However, the response of the species to three silvicultural treatments: CBS, regrowth thinning, and conversion of native forest to eucalypt plantation, is undergoing investigation in a before-after, control-impact study established in 1998 (Munks, Meggs and Richards unpubl. data). Hence the conservation strategy developed for this species represents an example of a fully-integrated adaptive management approach (Hopkins and Saunders 1987; Lindenmayer and Recher 1998), with future research results to be incorporated into the strategy as they become apparent.

As only broad habitat characteristics could be identified for the two *Lissotes* spp. and no 'hotspots' of abundance were found, the conservation of these species at the landscape-scale is largely dependent on existing large ecological reserves and the interconnecting system of wildlife habitat strips. For *L. latidens*, this is supplemented at the local or coupe scale by extra protection given to streamside reserves, including their extension to small headwater streams (Forest Practices Board 2001b). A blanket prescription for the retention of CWD at the local scale is recommended for the conservation of *L. menalcas*, but no minimum levels of CWD-retention are advised (Forest Practices Board 2001b). Additional mechanisms for the conservation of these species' habitats, based on the forest structural elements important for these species as identified in these studies, should be considered. Specifically, it is recommended that the utility of alternative silvicultural systems to CBS that include in-coupe retention of forest structural complexity, such as those currently being trialed in the wet eucalypt forests of southern Tasmania (Hickey *et al.* 2001), be investigated. This analysis will need to incorporate the requirement for recruitment of suitable habitat into the future. The inherent uncertainty involved in the further development of conservation strategies for

these two species, given the lack of data on the effects of habitat disturbance on the species, reinforces the need for the establishment of monitoring programs to assess the effectiveness of any conservation management strategy developed for threatened species (Munks *et al.* submitted).

Potential of threatened invertebrates as umbrella species in off-reserve areas

Conservation biology has traditionally focussed on the conservation requirements of single species, particularly large, charismatic vertebrates. With current estimates of global diversity in the order of 15 million species (Stork 1999), of which invertebrates comprise anywhere between 90-99% (Southwood 1978; Franklin 1993; Ponder and Lunney 1999), it is obvious that a species-by-species approach to biodiversity conservation is unsustainable. The majority of invertebrate species remain undescribed and even for many of those with names, we know little more than what they look like and where they were collected, which is often from only a single locality (Grove and Stork 2000). Even for one of the better known invertebrate orders such as the Coleoptera, it is suggested that three-quarters of a global estimate of 4 million beetle species have yet to be described, and that at the current rate of progress this could take another 200 years (Grove and Stork 2000). In the face of ongoing land-use practices that can result in the loss or alteration of habitats at a landscape scale, such as intensive forestry, there have been numerous calls for shortcuts to circumvent these impediments and ensure we achieve our goal of biodiversity conservation (eg. Bridgewater 1993; Franklin 1993; Hobbs 1994; Simberloff 1998; Noss 1999; Taylor and Doran 2001; Lindenmayer and Franklin 2002).

Proposed shortcuts for biodiversity conservation can be broadly grouped into two categories: variations on the single-species approach and landscape-scale ecosystem or habitat conservation. Variations on the single-species approach include the concepts of indicator, keystone, flagship and umbrella species (Simberloff 1998). The indicator species (or taxa) concept has proven very popular with forest managers as a possible tool to demonstrate ecologically sustainable forest management (eg. Michaels 1999; Taylor and Doran 2001). However, it has proven very difficult to demonstrate that a species' (or group of species) presence/absence is indicative of a range of other species (Prendergast *et al.* 1993), let alone that its response to habitat disturbance is

representative of those other species (eg. Franklin 1993; Margules *et al.* 1994; Michaels 1999).

Similar problems beset the keystone species concept as advocated by Simberloff (1998), which by definition is a variant of the indicator species concept (Yen and Butcher 1997). A keystone species is one that plays a critical ecological role and whose loss from a system will result in the loss of a suite of dependent species (Simberloff 1998). Identification of keystone species therefore requires an understanding of the ecological roles of a significant proportion of the component species of a system and the interactions between them. Conversely, flagship taxa are essentially removed from their ecological context. They are generally charismatic or 'poster' species that can be used to promote the concept of conservation and influence political debate. Well-known examples include koalas, whales and bilbies. New (1995) advocated butterflies as potential flagship invertebrate species, but given the generally poor public perception of invertebrates, there is likely to be a limited pool of other candidates in the short-term (Yen *et al.* 1990). Similarly, the umbrella species concept does not rely on any strong direct ecological link between species. An umbrella species is one whose conservation will indirectly result in the conservation of a suite of co-occurring species (Yen and Butcher 1997). This concept has grown from the untested belief that conserving wide-ranging, large vertebrates that may require relatively large areas of habitat will automatically conserve many other species. The Northern Spotted Owl has been held up as an example of such a species due to its requirement for large areas of old-growth forest (Simberloff 1998). However it has been questioned whether its conservation will also conserve the 6,000 invertebrate species also inhabiting these old-growth forests, because the precise habitat requirements of most are unknown (Lattin 1993). It is generally considered that invertebrates have limited value as umbrella species because of their perceived need for relatively small areas of habitat (New 1991). In Victoria, for example, one of the first major conservation efforts for an invertebrate, the Eltham Copper butterfly, focussed largely on only 3 ha (Yen *et al.* 1990).

It was beyond the scope of the studies presented here to test whether any of the three stag beetles constitute indicator or keystone species. However, it is possible to assess their potential as flagship or umbrella species for forest biodiversity conservation based on the conservation strategies developed for each species. Whilst fellow ecologists

have described *H. simsoni* as a 'sexy beast' (J. Meggs, pers. obs.), the general poor public perception of invertebrates (Yen *et al.* 1990) is likely to limit the usefulness of the three stag beetles as 'poster' species for biodiversity conservation, without an accompanying education campaign on the positive benefits of invertebrate conservation (New 1991). Nevertheless, *H. simsoni* has proven to be a valuable flagship species in being instrumental in raising the importance of a more strategic approach to the conservation of many threatened species by land managers (P. Wells, Forestry Tasmania, pers. comm.), as opposed to the general public. This species also exhibits potential as an umbrella species over a significant area of north-east Tasmania for an unknown number of co-occurring species also threatened by the conversion of native forest to plantation. Similarly, *L. latidens* can be considered an umbrella species for riparian species, particularly throughout the drier portions of its range where such areas may constitute a distinct habitat relative to upland areas. Of the three species, *L. menalcas* has probably exhibited the greatest potential as both a flagship and umbrella species for the conservation of CWD-dependent biodiversity. The importance of this habitat for forest biodiversity and the threats to its maintenance from intensive forestry is well-established overseas. Yet it can be argued that it was the coincidence of a significant proportion of the habitat of *L. menalcas* with areas identified as a potential source of fuelwood for a power station (Grove *et al.* 2002) that led to a recognition by forest managers in Tasmania of concerns regarding the ecological sustainability of this habitat under current and proposed forest management regimes. This recognition has subsequently been translated into action via the development of a research program focussing on CWD dynamics and its dependent biodiversity (S. Grove, pers. comm.). The true test of the value of this species as a flagship species for CWD-dependent biodiversity will be whether mitigation measures, developed primarily with *L. menalcas* in mind, are also applied in other areas of the State where CBS and/or fuelwood harvesting are proposed and no CWD-dependent threatened species are known to occur. Although *L. menalcas* appears to have a preference for a specific form of CWD (ie, large diameter logs in the intermediate stages of decay), the dynamic nature of this habitat and the need to ensure ongoing recruitment of CWD should capture habitat for a succession of species associated with different stages of the decay process (eg. Savely 1939; Wilson 1959; Elton 1966; Greenslade 1972; Esseen *et al.* 1992), including vertebrates (Lindenmayer *et al.* 2002).

The major limitation of the umbrella species concept as a means to capture forest biodiversity is that its usefulness is restricted to co-occurring, ecologically similar species (Simberloff 1998; Mesibov *et al.* 2002). Hence, a reliance on threatened species to act as umbrella species could be considered an extension of the set-aside approach to biodiversity conservation, where mitigation measures are restricted to specific portions of the landscape and associated species (Franklin 1993). Mesibov *et al.* (2002) discuss the inefficiency of this predominantly 'single species' and localised approach to the conservation of Tasmania's invertebrate diversity in general. Instead, they advocate a functional group and biodiversity 'hotspot' approach through the intensive sampling of invertebrates in areas most obviously at risk from proposed land-use activities. This would undoubtedly lead to an improved knowledge of the occurrence of many poorly known and currently undescribed species in specific areas. However, it is difficult to see how in the short-term this may be translated into conservation measures without the landscape context provided by knowledge of biodiversity 'coolspots' or knowledge of the distributions, habitat characteristics and responses to disturbance of component species. It is also evident from the present investigation of the three stag beetles that apparently similar species may not easily fit into our current understanding of functional groups.

Despite the potential for threatened invertebrates to function as umbrella species for habitats at risk from off-reserve land-use practices, a species-by-species approach to forest biodiversity conservation, focussed on the listing of individual invertebrate species, is unsustainable. As noted in Chapter 3, since the proclamation of the TSPA in 1995, a further 21 invertebrate species have been added to the schedules of the Act and only one invertebrate species has been delisted; this, despite the significant expansion of the CAR reserve system in the interim. Through sheer weight of numbers, it is likely that the more we know about our invertebrate fauna, the more species will be listed, rendering the threatened species management system unworkable. However, until alternative or additional mechanisms are available for the conservation of important invertebrate habitats across the landscape, it is apparent from the studies presented here that threatened species management can be of great value in 'off-reserve' areas, particularly in landscapes where an extensive reserve system is already in place.

Alternatives to the single-species approach to biodiversity conservation

As many of the single-species shortcuts proposed suffer from the same knowledge impediments, which will take some time to overcome, many authors have advocated a habitat or ecosystem approach as a surrogate or umbrella for biodiversity conservation (eg. Bridgewater 1993; Franklin 1993; Hobbs 1994; Noss 1999; Lindenmayer and Franklin 2002). The main criterion used currently in Australia to assess and classify ecosystems is the dominant floral communities (Yen and Butcher 1997). In Tasmania, a somewhat more refined approach was used in a review of the representation of forest ecosystems in protected areas (Working Group for Forest Conservation 1990). This approach incorporated altitudinal variation and geology into the traditional floristic classification of forests, and ultimately led to the addition of 400,000 ha of forest to the existing reserve system (Commonwealth of Australia and State of Tasmania 1997). Invertebrates were not part of the ecosystem classification or reserve selection process, yet the ultimate test of the utility of such areas in conserving biodiversity will be how well they have captured populations and habitats important to invertebrates, particularly those habitats vulnerable to off-reserve land-use practices. Due to the relatively large size of these areas, they will inevitably have succeeded in achieving this goal to some extent. Again, however, this has to be taken as a matter of faith as no comprehensive inventory of the invertebrate fauna captured within the reserves has ever been conducted. It was clear in the studies presented here that the distributions and abundance of the three stag beetles showed no association with forest community species composition. Indeed, there is a growing body of evidence that the distributions of many invertebrate species and communities do not track the distributions of floristic communities (eg. Gardner 1991; Taylor *et al.* 1994; Sanderson *et al.* 1995; York 1999; but see Yen 1987).

The maintenance of habitat across many spatial scales is paramount in the conservation of biodiversity (Lindenmayer and Franklin 2002). The loss of habitat is a key driver of species loss (Fahrig 1999). Hence, a habitat approach has been advocated as a possible focus for biodiversity conservation in the face of uncertainty (eg. Yen *et al.* 1990; Yen and Butcher 1997; Taylor and Doran 2001; Lindenmayer and Franklin 2002). It is relatively easy, at least in theory, to identify and protect highly localised and readily identifiable habitats such as caves or swamps (Yen and Butcher 1997). However, many of the habitats known to be important to invertebrates such as soil, leaf litter and coarse

woody debris (CWD) are relatively ubiquitous in natural forests and hence difficult to prioritise for protection in any spatially explicit way without knowledge of their component fauna, particularly for those species sensitive to habitat disturbance. In the face of limited knowledge and much uncertainty, the conservation of such habitats important to invertebrates requires an approach that ensures their maintenance throughout the forest landscape (Franklin 1993; Lindenmayer and Franklin 2002).

A risk-spreading approach focussed on the maintenance at multiple scales of habitats known to be at threat, as advocated by Lindenmayer and Franklin (2002) is likely to be the most productive avenue for the conservation of forest-dependent invertebrates. Whilst policy makers have embraced the concepts of biodiversity conservation (DPIWE 2001) and ecologically sustainable forest management (Commonwealth of Australia and State of Tasmania 1997), there appears to be little appreciation of the predominance of invertebrates in forest ecosystems and hence what is actually required for the goal of biodiversity conservation to be met. This is evidenced in the relatively narrow focus of the draft *Tasmanian Nature Conservation Strategy* (DPIWE 2001), which largely relies on existing mechanisms for conservation, such as set-asides and threatened species management rather than emphasising the importance of habitat maintenance across the landscape. This lack of mechanisms for biodiversity conservation beyond set-asides and threatened species management becomes apparent in a scenario that assumes that the conservation strategies recommended for the three stag beetles ensured their security in perpetuity. Accordingly the species should be delisted as the specific threatening processes are no longer operating. However, there would be no enforceable mechanism to ensure appropriate management of the species habitat into the future. Similarly, monitoring of the ecological sustainability of forest management in terms of biodiversity conservation is limited to measuring trends in the amount of RFA forest communities and the population trends of a select group of common and threatened vertebrate species (RPDC 2002). Implicit in this narrow view of biodiversity conservation, and the lack of off-reserve conservation mechanisms explicitly targeted at invertebrates, is an acceptance of biodiversity loss in significant portions of the landscape because we do not know what is there. Yet, as argued by Taylor and Doran (2001), we can identify key invertebrate microhabitats such as soil and litter, foliage and canopy, bark and branch, dead standing trees and CWD that can be targeted for research and monitoring, particularly in the context of those components of their invertebrate

fauna restricted to late-successional and old-growth stages. Under current forest management regimes, it is these growth stages that are likely to be under-represented in the production forest landscape (Franklin 1993; Lindenmayer and Franklin 2002). Whether truly 'representative' or indicator species can be identified for these habitat/growth-stage complexes is questionable given the range of responses of invertebrates to habitat disturbance (Friend 1995), but is worth investigating. In the interim, threatened invertebrates may fulfil the role of representative species, at least over a portion of the off-reserve landscape, as they generally represent species that we know are associated with habitats under threat, albeit drawn from the tiny fraction of forest biodiversity that we know anything about.

Regardless of whether a single-species, a habitat/ecosystem approach, or a combination of approaches, is taken to the conservation of forest biodiversity, if we are to have any confidence in broad-scale strategies for the conservation of invertebrates, they will need to be under-pinned by a knowledge of the individual habitat and conservation requirements of at least a portion of the component fauna. Similarly, if monitoring of the ecologically sustainable management of our forests is really meant to be some measure of our ability to achieve biodiversity conservation, then monitoring of populations of well-known invertebrates must be included. It is important, therefore, that in our pursuit of shortcuts for biodiversity conservation, resources continue to be provided for research aimed at obtaining detailed distribution and habitat utilisation data of individual species, such as that presented here. Ultimately, it is the response of individual species to our land management activities that provides the measure of the degree to which we have achieved our goal of conserving forest biodiversity.

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